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Procedures Manual for Ground Water Monitoring at Solid Waste Disposal Facilities

Wehran Engineering Corp., Middletown, NY

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1.0 EXECUTIVE REVIEW

1.1 INTRODUCTION

This manual brings into one volume, information valuable as a reference source for those persons actively engaged in or planning ground water monitoring programs at solid waste disposal facilities. It was completed prior to passage of the Resource Conservation and Recovery Act of 1976, which contains major provisions to move the country more rapidly toward environmentally safe solid waste disposal practices. The implications for monitoring activities are clearly great, and this manual should serve as a particularly useful tool as State solid waste agencies proceed to strengthen their land protection programs. The manual is primarily addressed to the supervisory personnel of solid waste regulatory agencies, although its contents can be readily used by engineers in the field. It is offered as a guide to be used and tailored by the supervisory personnel at their discretion and guidance to persons without prior training or experience. It should prove helpful to the operators and managers of solid waste disposal facilities who find a need for a familiarity with and understanding of the fundamental principles involved in ground water pollution and monitoring.

Generally, this manual includes fundamentals and provides guidance to assist the user in:

- establishing the need for monitoring;
- assigning priorities for facilities to be monitored;
- implementing and directing cost-effective, on-going monitoring program responsible to the purposes and data needs established.

The information, as presented, is offered as guidance and suggested methods only. Site specificity is recognized throughout the manual.

The purpose of the Executive Review is to provide the reader with an overview and summary of the contents of the manual. Background information on the impact of waste disposal on ground water can be found in "The Report to Congress: Waste Disposal Practices and Their Effects on Ground Water" (January 1977, EPA Office of Water Supply and Office of Solid Waste).

1.2 PURPOSES AND OBJECTIVES OF MONITORING

The very fundamental objective of monitoring land disposal sites is to serve as a check on potential leachate contamination. There are two basic land disposal situations which require distinct approaches to monitoring, existing sites and new sites. At existing sites where ground water contamination has already begun, it is important to protect the users (present and future) of the ground water. At new sites and at existing sites where contamination has not occurred, protection of the ground water resource may receive additional emphasis.

Where a potential for leachate generation exists, an assessment of the leachate problem must be made to determine the control strategies for a particular land disposal site. If the natural hydrogeological conditions of the site cannot by themselves provide adequate control, artificial leachate control techniques (e.g., plastic liners, clay liner) may be needed. Ground water monitoring will play an all-important role in making such an assessment and in providing long-term verification.

Several purposes of monitoring can be enumerated.

- Monitoring where the control strategy is the protection of the ground water user.
- Monitoring where the control strategy emphasizes additionally the protection of the ground water resource.
- Collection of evidential data for enforcement purposes.
- Demonstration of the presence or probable absence of leachate contamination as may be done in a statewide evaluation of the extent of the problem.
- Check on the effectiveness of an engineering design and site preparation including such items as liners, berms, subsurface diversionary measures, etc.
- Collection of prescriptive monitoring data in response to descriptive monitoring to develop effective engineering remedies for contamination problems.
- Performance of scientific investigations in developing and validating design criteria, such as: rates of leachate movement, attenuation, etc.

Monitoring for each purpose differs in both scientific and economic scope. For example, a scientific investigation would attempt to quantify as much as possible the amounts, movement, and attenuation of leachate contaminants and their impact on the environment. Such a program requires very extensive monitoring consisting of tens of monitoring wells and the performance of extensive analysis with frequent sampling. It would also involve extensive hydrogeologic and geophysical study with detailed field measurements of soils and ground water flows. On the other hand, the simplest monitoring program for determining the presence or probable absence of contamination, such as in a statewide survey, might only require one or two wells. Perhaps a detailed site inspection by an experienced hydrogeologist familar with the geology would be sufficient. Obviously, these extremes would differ in effort and cost by one or two orders of magnitude ($100,000 vs. $1,000).

Generally, most monitoring programs will fall between these two extremes. Monitoring programs may evolve in a sequential pursuit of the various purposes listed above. For example, the presence/probable absence monitoring will almost always be the starting point from which to build a program. Once this has been satisfactorily addressed, the circumstances specific to the site and State code will dictate the next step. If there are nearby water supply wells which are potentially threatened by leachate contamination, immediate steps would be taken to check their quality. If there is no real threat to public health but the State's control strategy is resource protection, immediate steps would be taken to determine if leachate contamination is entering the groundwater. In the former case, monitoring would initially focus on the nearby water supply well; in the latter, monitoring would focus on the landfill.

This might logically lead in a much more comprehensive program to gather evidential data for an enforcement case. In almost all cases, evidence gathering will require a more intensive monitoring effort consisting of additional sampling points and analyses to remove any doubt that the landfill is the source of contamination.

Additional monitoring may be required to prescribe engineering design criteria; and upon implementation, a check might be made of the effectiveness of the said design. For example, several additional sampling points may be established at the landfill in order to more accurately measure the dimensions of the leachate plume (body of leachate-contaminated ground water) as well as its movement, dispersion, and attenuation characteristics. This data will provide valuable input into the engineering of the remedial program.

An example of additional monitoring to check the effectiveness of a remedial program would be the installation of sampling devices just below an underliner or outside a cut-off wall to detect any pass-through of contaminants.

Following is a brief summary of postulants, facts, and judgements gleaned through the knowledge and experience of State solid waste management officials which will put the manual into perspective.

- At the present time, extensive use of ground water monitoring as a part of solid waste regulation is not being made.

- Where monitoring is required of landfill operators, seldom is the data used in litigation for lesser enforcement actions.

- State solid waste management administrators foresee extensive use of ground water monitoring in the future and would welcome guidelines for its use. Where ground water monitoring is now required, it is almost exclusively for new sites.

- The statutes of many States would authorize ground water monitoring by defining waters of the state to include ground water as well as surface waters.

- Usually there are no regulations of specific applicability to monitoring wells or specifications as to their construction. Where wells are required, it is done on a case by case basis.

- Agreed upon standards defining leachate pollution do not exist; some States use Public Health Service Potable Water Standards while others use the guidelines of the National Water Well Association and some operate under the statutory principle of absolute nondegradation.

- The key to adequate control of landfills is a permit system. Requirements for both existing and new landfills are imposed as conditions of permits and monitoring results may be keyed to permit conditions.

- Some express the opinion that rather than contaminate ground water, discharge areas such as flood plains should be employed for landfills on the theory that it is better to have the leachate travel into the ocean then irrevocably into the ground water. Others feel that if aquifers were contained and are not in use for potable supplies, the site should be continued as a waste sink with no necessity for monitoring.

- In some locations, ground water pollution cases or denial of permits have been done exclusively on the basis of geological information without any monitoring.

- There appears to be a general reluctance to impose ground water monitoring requirements on publicly-operated landfills due to the cost factors.

1.3 TYPES AND LIMITATIONS OF MONITORING

It is important for the administrator to have an understanding and appreciation for what monitoring can and cannot do--the kinds of information that it can provide as well as its limitations. Too often monitoring is arbitrarily approached, producing insufficient, inaccurate, or unnecessary information for the purpose intended. In order to avoid this and to gain a fuller appreciation of monitoring, it is helpful to categorize the process into two fundamental levels:

- detective monitoring, which establishes presence/absence of contaminants and the need for further monitoring, and

- interpretative monitoring, which determines extent of damage and prescribes remedial action.

In the effective administration of a detective monitoring program, it is important to have some appreciation of hydrogeology and its limiting influences on monitoring. Monitoring ground water at a land disposal site is completely different from monitoring a surface water, a waste discharge pipe, or an air-pollution plume. The main differences are the unpredictability and the difficulty and expense in obtaining representative data. For example, modeling has been successfully used in predicting the impact of a waste discharge on a stream or of a stack emission on the air. However, for ground water, modeling is used in the sense of constructing a model of the system as it exists, i.e., gathering enough data to enable a three-dimentional image to be made of the plume. While modeling does not necessarily imply the development of a mathematical simulation for use as a predictive tool, it does require the collection of a considerable amount of data over a long period of time--more than would be practical for most land disposal sites or regulatory program needs. For example, the following types of data that such modeling might include demonstrate this point:

- Geologic:
  - surface geology (topography and type/depth of overburden)
  - lithology of aquifer
  - type of geologic formation (local stratigraphy and structure)
- Hydrogeologic:
  - depth to water table
  - water-table contours
  - thickness of aquifer(s)
  - relative hydraulic heads, if more than one aquifer
  - annual precipitation
  - aquifer permeability and porosity
- Geochemical:
  - background-water quality
  - chemistry of geologic formation
  - presence of other sources of chemical or biological contamination.

It is very difficult and expensive to obtain representative data necessary to support math modeling which in its present pioneering state provides limited reliability. While it may have application in highly critical situations, such as detailed scientific investigations of hazardous or toxic material threat to drinking supplies, it is not yet appropriate for general use in State programs.

The many geohydrologic limitations inherent to ground water monitoring require much interpolation and scientific estimation on its use. It is difficult to obtain samples at many points. Unlike the situation found in streams or in the air, ground water moves very slowly and there is little mixing. Thus, there may be a wide range of unpredictable variations in contaminant concentrations within a plume of leachate-enriched ground water. Several limitations as they relate to ground water monitoring, are discussed below:

- A monitoring well is sampling a very small part (point) of the aquifer, thereby limiting its representiveness to the quality of the ground water in the immediate area of that well. Interpolation of the point data is compounded by the hydrogeologic characteristics in leachate movement that produce wide variations over short distances and time periods.

- It is expensive to obtain reliable water-quality data at chosen points in an aquifer. Unlike surface water where a sample can be arbitrarily taken at any point, moving the sampling point in ground water implies the installation of additional monitoring wells.

- Sample extraction may be made difficult due to the tightness of a geological formation or the depth to the ground water.

- Determination of the flow rate and direction of ground water are prerequisites to monitoring well placement. Drilling will be required and measurements must be made of the piezometric surface.

- Ground water flow rates are extremely slow resulting in a correspondingly slow change in its quality at a particular monitoring well. This phenomena will, in most cases, require data collected over long periods of time (years or months) to perform a comprehensive analysis of the landfill. There may even be situations such as fractured rock conditions which are so unpredictable as to frustrate a very intensive monitoring effort.

1.4 FATE OF INDICATORS

Many of the chemical components of leachate are common to all municipal waste sites. Other components such as certain heavy metals (Cr, Ni, Cd) or synthetic organic compounds (pesticides, PCB's, solvents) may be contributed by wastes specific to a site. The dissolved chemicals leaving the bottom of the solid waste zone move with percolating water with different degrees of interaction with each other and with soil or sediment components.

1.4.1 Contaminants Not Usually Considered Toxic

Inorganic components associated with leachate include the cations (positive charge), sodium, potassium, calcium, magnesium, iron, manganese, and ammonia. Anions (negative charge) include chloride, sulfate, phosphate, and bicarbonate. Cations are subject to attenuation by cation exchange reactions on adsorptive surfaces in soils and sediments.

Anions, except for phosphate, are not appreciably affected by sorption reactions. Thus, they move with percolating water with little change in concentration. Ammonia is oxidized to nitrate under aerobic conditions. Therefore, ammonia may be present in leachate as it leaves the surface; and by the time it reaches ground water, significant conversion to nitrate would take place.

Bicarbonate, chloride, and sodium are frequently the ions of choice for tracing leachate in ground water. The choice of these, or others, depends upon the quality of ambient ground water in the area. The ion of choice must be clearly enriched in respect to background-water quality.

1.4.2 Contaminants Usually Considered Toxic

The suite of heavy metals, boron, selenium, arsenic, and nitrate are the inorganic chemical species identified with varying degrees of toxicity. These constituents are present in leachate in concentrations which more directly reflect the specific landfill than do the concentrations of the preceding constituents.

Most of the heavy metals are subject to attenuation from ion exhange or precipitation reactions. Boron, selenium, arsenic, and nitrate occur as anions slightly affected by sorption processes.

Since few organic constituents have been positively identified, the toxicity is difficult to assess. From the usual odor associated with leachate, organic constituents would be considered decidedly undesirable even if not proven clinically toxic. Organic constituents are the decomposition products (largely anaerobic) of a variety of organic wastes deposited in the landfill. If water-soluble organic matter is deposited, it may also be present unaltered. Organic attenuation occurs from a limited amount of adsorption and microbial degradation occurring in the zone of aeration and possibly continuing in the aquifer.

Because of their virtual absence in ambient ground water, some of the toxic components may make excellent leachate tracers. For example, borate, a conservative ion and an unubiquitous component of ground water, has been reported in leachate. These qualities make it a useful tracer where present.

1.4.3 Geohydrology

A few hydrologic principles should be recognized in dealing with leachate plumes in ground water:

- Leachate is not diluted with the entire body of ground water but tends to remain as an intact body with only slight dispersion and diffusion along the edges.

- Leachate constituents actually move faster than the average ground water velocity because of hydrodynamic dispersion.

- The path of a leachate plume will follow the direction of ground water flow. Diversions in flow direction from induced changes in gradient (e.g., a pumping well) will also divert the leachate plume.

- Hydraulic and lithologic conditions and leachate density determine the vertical depth to which leachate will migrate into the aquifer. The thickness of the plume in the aquifer will probably increase with distance downgradient from the source.

- The extent and movement of various constituents in the plume will vary due to the effects of partial attentuation.

1.5 COST OF MONITORING

Costs will vary directly with the purpose and therefore the level of the monitoring program. The more information desired and the more quantitative the site evaluation, the higher the costs will be. A few examples will serve to demonstrate this point:

- If a state solid waste program were to assess the extent of leachate contamination in the state, the emphasis would be on interpretative monitoring to perform a qualitative assessment of the problem and to minimize hard data gathering by drilling borings, wells, etc. Any sampling and analysis done would be limited to existing wells (on or off site), leachate springs, or surface waters and analyzed for key indicator parameters only. Such an assessment program for an individual site would cost anywhere from a few hundred dollars to a few thousand dollars depending upon the magnitude of a site and its potential threat to public health and safety.

- Another important purpose of monitoring is enforcement. In almost all cases costs of enforcement monitoring will be much higher than for assessment monitoring as given in the first example. Quantitative data will be necessary on the leachate contamination at the site. There must be sufficient data to show a preponderance of evidence or to prove beyond any reasonable doubt that contamination exceeds the applicable standards and that the excess contamination is caused by the land disposal site. The costs of enforcement monitoring for ground water contamination cases will range from several thousand dollars to several tens of thousands of dollars. The strength and specificity of the ground water protection laws and the site conditions are the main causes of cost variations. For example, monitoring for zero-discharge laws would require sampling devices beneath or immediately adjacent to the downgradient landfill edge. More costly monitoring would be involved in those areas where non-degradation or "property line" laws are in effect. In these cases, several monitoring wells at various distances and depths downgradient as well as comprehensive conditions, especially depth to ground water, will directly affect the installation costs of sampling devices. The number of analytical parameters required by regulations will have a direct effect on the analytical costs.

- Another important purpose of monitoring is the performance of scientific investigations to develop and verify design criteria and standards. Such investigations involve comprehensive surveillance of the disposal site including thorough analysis of site conditions, blanketing the site with numerous sampling devices along a horizontal and vertical profile, and the running of frequent and extensive analyses. The entire program will require a team of experts. Costs ranging from several tens of thousands of dollars to a few hundred thousand dollars would not be unusual.

General cost estimates are presented throughout the manual for order of magnitude and comparative purposes only.

1.6 IMPLEMENTATION OF AN EFFECTIVE PROGRAM

In order to establish an effective state program, it must proceed simultaneously on enforcement and locating alternatives with a systematic approach and a good data base. The state must possess a working knowledge of the hydrogeological setting and the probable extent of contaminant transport at all existing landfill sites.

1.6.1 Assessment of Existing Land Disposal Sites

The assessment of existing sites must provide sufficient information concerning the probable contamination profile at each major site. It must be approached as a long-range continuing effort. It requires staff geological talent (perhaps one experienced and two junior geologists for a medium-sized northeastern state) and will commence with three to five mandays per site including search of office files; U.S. Geological Survey, SCS, and other files; and field visits for an initial assessment. Maximum and minimum potential plumes should be sketched in cartoon form (Chapter 2). Upon completion of a significant number of sites, geophysical and field analysis will begin at those more likely to pollute. Well monitoring at existing sites will probably not be indicated prior to this point in program maturity and only in a very few sites initially upon completion of geophysical assessment. Pragmatic resource limitations such as the size of the enforcement program staff will constrain the rate of data collection. No administrator wants ten new major violations reported all at once. The purposes for monitoring are not limited to the support of enforcement actions. Other purposes include:

- To assess empirically the state's facility standards on State geology under local rainfall with local solid waste in place.

- To provide justification for appropriate staffing and budgeting in order to handle actual workload.

- To provide sufficient information to make realistic estimates of the best least cost exit from a site and the environmental consequences of delays.

- To provide bases for state-wide facility scheduling for optimal disposal resource allocation.

1.6.2. Review and Orientation of Present State Program

Simultaneously with the site assessment activity, the existing monitoring program should be reviewed. This might include such tasks as the following:

- Listing of all sites with established monitoring points (surface or ground water), including sites presently being studied for pollution abatement and/or permitting.
- Continuing to require monitoring a new or potential serious contaminating site with closer State input on location, design, etc.
- Revising organizational water analyses records to be more consistent with future programs.
- Establishing a monitoring station numbering system with field markers.
- Investigating organizational records and listing chemical and physical indicators of leachate contamination from normal solid waste and industrial waste. This should curtail lab time per sample.
- Running water analyses of discharges from existing special waste sites to aid in identifying indicators representative of these leachates.
- Adopting recommended ground water and soils data-collection methods and issuing a manual on the same.
- Arranging interviews with major State drilling contractors/ground water consultants to discuss proper sampling methods and procedures. Contacting universities and research groups regarding possible research projects on the environmental effects of leachate and the development of design criteria and standards.
- Locating on up-dated topographic maps all existing sites, monitoring locations, documented areas of contamination, potential areas of contamination, etc.
- Placing more emphasis on geophysical work, especially during the subsurface investigative stage.
- Purchasing of equipment such as pumps for shallow well sampling, deep well sampling, conductivity meters, pH and temperature probes, individual field kits, etc.
- Determining priorities of monitoring at waste disposal sites.
- Determining categories of enforcement action based upon monitoring results.

- Investigating computerization of analyses and of sampling requests

- Correlating leachate quality and quantity to seasonal patterns.

1.6.3 Establish Monitoring Networks and Data Needs for Land Disposal Sites

Utilizing the background assessment information obtained in Section 1.6.1, priorities should be assigned to sites relative to the need for monitoring. The state will then work with the landfill operator in establishing the number and placement of monitoring wells or other sampling devices. Listing of analyses to be performed as well as sampling schedules must also be agreed upon at this time.

1.7 STEP OUTLINE OF MONITORING PROCEDURES

For every landfill, both existing and proposed, the establishment of a ground water monitoring program should be investigated. Although the use of a "cookbook" approach cannot be justified, the establishment of a monitoring program can be described in a logical sequence of individual steps. However, as with any complex situation, the best possible solutions will require incorporating original thought during each step.

Design and implementation of a monitoring program is usually accomplished by the combined talents of a civil environmental engineer and hydrogeologist. The hydrogeologist provides the expertise needed to interpret the subsurface conditions and ground water flow patterns, and the engineer applies the subsurface data in the landfill design and its pollution control features.

The steps presented in this chapter are intended to indicate the logical progression of required efforts and, therefore, are not accompanied by detailed descriptions. Such descriptions can be found in other chapters of this manual, or in the references cited at the end of the appropriate chapter. The general approach consists of review of existing information and employment of non-sampling field inspection techniques and sampling techniques.

1.7.1 Step I--Initial Site Inspection

All information is gathered from an inspection of the landfill, examination of landfill records and other existing information such as topographic maps, and discussion with landfill-operating personnel. The purpose is to define, with a minimum expenditure of time and money, the probable magnitude of the ground water contamination problem. This will determine the urgency for conducting a detailed study and establishing a monitoring program.

1.7.1.1 Nature of the Solid Waste. The nature of solid waste disposal varies widely from landfill to landfill depending largely on the types of waste generated in the area, the regulatory agency for the area, the landfill operator, and economics. A determination of the types of wastes accepted at a particular landfill(both currently and historically) is critical to the monitoring evaluation.

The categories which are accepted at the landfill should be determined; and in the case of industrial wastes, more detailed information should be sought.

1.7.1.2 Aerial Extent and Thickness of the Landfill. The pollutant application rate, represented by the size and thickness of a landfill is an important factor in estimating the volume of leachate generated as well as the concentration of contaminants in the leachate.

1.7.1.3 Pretreatment and In-Place Treatment of Solid Waste. In most cases, solid waste is compacted using heavy equipment after it has been placed in the landfill. The compaction should be determined to estimate the density and field capacity of the landfill. Shredding and baling of solid waste will increase its density. Incineration and resource recovery operations will alter the composition of solid waste, and consequently, the nature of the leachate generated.

1.7.1.4 Landfilling Procedures. The procedures used in placing and covering the solid waste at the landfill site will influence the volumes and characteristics of the leachate generated. Other important factors to be considered are the different types of solid waste, the thickness of the cover layers and the solid waste between them, and the type of material used for the cover.

1.7.1.5 Rate of Landfilling and Age of Solid Waste. Since a thick section of solid waste can absorb more water (field capacity), the rate at which the thickness of a landfill increases will affect the volume of leachate generated. If the landfill thickness increases at a sufficient rate relative to precipitation and upon completion is covered to exclude precipitation, very little leachate will be generated. Thus, the rate of filling will influence the design of a monitoring system.

Solid waste layers of different ages produce leachates of different chemical characteristics. This factor may be useful in designing a monitoring program; however, other factors such as solid waste composition have a more noticeable influence on leachate than does age.

1.7.1.6 Liners and Covers. If a landfill is equipped with an underliner and a leachate collection system, ground water is assumed to be uncontaminated. A monitoring system would then be designed only to test the validity of this assumption. Similarly, if the landfill is completed and covered to prevent the infiltration of precipitation, initial monitoring would be necessary only to establish the absence of contamination. If initial monitoring data show the liner or cover system to be ineffective, an expanded monitoring program would be designed to define the extent of the problem and the necessary corrective measures.

If the bottom liner and collection system are effective, the volume of leachate actually collected should be equal to the predicted volume of leachate being generated. Relative to the effectiveness of covers and surface drainage systems, runoff and evapotranspiration account for a large percentage of precipitation. Factors such as cover permeability, slopes, and vegetation-type would be considered in this determination.

1.7.1.7 Visual Survey of Topography and Geology. The primary purpose of this effort is to establish estimates of surface runoff and infiltration patterns and to determine the general direction of ground water flow. The topography of the areas surrounding the landfill will establish the direction of surface-water flow, either toward or away from the landfill surface. Once site recharge and discharge areas are determined, the general direction of the ground water flow can be approximated.

1.7.1.8 Ground Water Use. Checks should be made into ground water use in the vicinity of the landfill.

1.7.2 Step 2 -- Preliminary Investigations

Once the need for a detailed ground water investigation and monitoring program has been established (Step 1), the program should be carefully planned. To accomplish this task efficiently, all existing pertinent data are gathered and examined at the outset. This data includes all information from Step 1 and any useful data available from outside sources. In addition, information which can be readily obtained at the landfill site but was not needed during Step 1 is gathered.

1.7.2.1 Existing Data. Necessary information other than that gathered in Step 1 includes:

- historical precipitation records for the site or a nearby area;
- geologic and topographic maps including the landfill site;
- geologic logs of any existing wells or test borings at or near the site;
- a recent aerial photograph of the site from which to prepare an accurate base map.

If potential sources of contamination other than the landfill are located in the vicinity, all available information regarding these sources (type and volume of waste, methods of disposal, etc.) also should be collected and reviewed.

1.7.2.2 Preliminary Site Investigation. Additional data which will improve the efficiency of a hydrogeologic investigation of a landfill (Step 3) include:

- analyses of water samples from surface water bodies and existing wells located on or near the site;
- analyses of samples of leachate from surface seeps;

- examination of site vegetation by a botanist for signs of stress;
- observations of surface drainage patterns during a rainfall;
- a check of building basements and other subsurface structures at the site for landfill gas accumulations.

Preliminary site investigation is not limited to supply wells. Surface-water use in the area must also be considered. Nearby surface water may be used for potable supply, fishing or shellfishing, swimming or other recreation, or wildlife habitat. Since surface-water bodies are often discharge areas, they are subject to contamination from leachate in the ground water system. Information should be established as to the location, use, and rate of flow for all surface-water bodies in the vicinity of a landfill. Natural water quality, existing contamination, and sources of contamination should be investigated. At discharge points, ground-water contamination becomes most conspicuous. Therefore, sampling of surface-water bodies may form an important part of a monitoring program.

1.7.3 Step 3 -- Definition of the Hydrogeologic Setting

The hydrogeologic setting of the landfill is probably the most important factor in establishing the need for and design of a landfill monitoring system. Prior to selecting a landfill site, information as to surficial and bedrock geology, depth to the water table, and direction and rate of ground water flow should be determined. In the past, subsurface conditions have not been well-defined; landfills have been located on lands such as swamps or abandoned gravel pits, traditionally considered useless and of low economic value. Ground water pollution potential is present in these areas; and in the case of gravel pits, the potential is high. For these reasons, the need for monitoring and abatement procedures is acute.

1.7.3.1 Surficial Geology. The purpose of a surficial geology survey is to establish the areal extent and thickness of the layers of the various types of deposits under and adjacent to the landfill. Also, the permeabilities and interconnections of these layers should be determined. The survey can be divided into three sequential parts:

- a review of geologic data gathered during Steps 1 and 2;
- geophysical surveys designed to complete subsurface information;
- test drilling to provide direct control for the geophysics, obtain more precise data in critical areas, and allow detailed analyses of the geologic samples.

1.7.3.2 Bedrock Geology. In some cases, bedrock will act as a barrier to leachate movement; yet in others, leachate may move into bedrock aquifers. The type of rock beneath the sites and the amount of fracturing will determine the role of bedrock in the movement of leachate. Determination of bedrock geology essentially will follow the steps outlined for surficial geology.

1.7.3.3 Ground Water. The ground water investigation should provide information regarding the following:

- depth to the water table;
- the extent of ground water mounding caused by an existing landfill;
- the natural rate and direction of flow;
- the degree of influence the landfill has on the rate and direction of flow;
- the locations of recharge and discharge areas;
- the types and interconnection of aquifers;
- the rate of site infiltration relative to the total ground water flow.

Much of this information would be obtained during and immediately following the geologic investigation previously outlined. During test drilling, data regarding water levels and head differences with increasing depth would be recorded. Test borings can be equipped with screens and test pumped at various intervals while other borings could be implemented as observation wells to establish aquifer characteristics and interconnection between aquifers.

Information relative to recharge at the site can be obtained from historical precipitation records and estimation of the surface runoff and evapotranspiration. These data are necessary to predict the probable size and shape of the leachate plume. Ground water recharge and discharge areas are determined by the results received from observing surface features and the relative water levels in observation wells.

A monitoring program is dependent upon the information gained from hydrogeologic investigations. The existing potential for a landfill to contaminate an aquifer or surface-water body will require the construction of a monitoring program. Information regarding the actual condition of threatened and contaminated aquifers (depth and extent) will aid in determining the most favorable locations and depths of monitoring wells. The size and complexity of a monitoring program will be based partially on the calculated volume of recharge throughout the landfill as well as the volume and rate of ground water flow. Subsequent steps outlined in this chapter will provide the necessary data for the refinement of the monitoring program initially outlined and for the elimination of its less important features.

1.7.3.4 Determination of Existing Water Quality. An accurate and complete record of existing ground and surface-water quality is necessary as background water quality in a monitoring program. If previous contamination has occurred, water samples should be collected from uncontaminated areas and analyzed to establish natural water quality. If sources of contamination are present (other than the landfill), the effects of these sources should be determined. Since the objective of a monitoring program is to determine change, historical data are indeed vital.

If outside sources of contamination are present (determined in Step 2) and if existing information is insufficient to define the problem, additional investigation would be necessary. Such an investigation would include:

- the rate of contaminant generation;
- the nature of the contaminants;
- the resulting degree of ground water degradation.

The monitoring system must then be designed to account for these "outside" contaminants. If not accounted for, these contaminants might be inadvertently attributed to the landfill.

Determination of the existing water quality is not limited to consideration of the primary contamination directly introduced. When landfill leachate reaches and blends with ground water, secondary reactions may occur. For example, the oxidation potential of the leachate-enriched ground water may be lowered by the mixing in of chemically-reduced leachate. This, in turn, may reduce and dissolve iron or manganese occurring as coatings in the aquifer materials. Cation exchange reactions releasing calcium and magnesium, changes in pH, or the precipitation of some leachate constituents are other reactions which could occur and change water quality.

1.7.3.5 Determination of the Rate of Leachate Generation. The leachate generation rate is determined by a water-balance study of the landfill and will influence the magnitude of the monitoring program. Necessary data for water-balance determination include: precipitaton data, landfill surface characteristics, vegetation-type and density, landfill site topography, ground water underflow rate, the rate of landfilling, and pretreatment and compaction of solid waste. A discussion of water-balance calculations is given in Chapter 5 of this manual.

1.7.4 Step 4 -- Determination of the Pollution Potential of the Landfill

The extent and design of a monitoring system will be influenced greatly by the pollution potential of the landfill. Estimation of the pollution potential essentially is derived by consolidation of all data gathered in Steps 1, 2, and 3. Determinations made would include:

- the location, size, and rate of the movement of the contaminated plume;
- the aquifers now affected and those possibly affected in the future;
- the types of contaminants present;
- and the degree of attenuation of those contaminants by the subsurface sediments.

These data can then be used to predict the total pollution damage that may be caused by the landfill if no action is taken. They also may be used to estimate the influence of various possible abatement procedures. Monitoring program data are then implemented in establishing the accuracy of these predictions and provide a warning when abatement systems are ineffective.

1.7.5 Establishing the Monitoring Program

The next step would be to compile the information gathered in the previous steps into a detailed written report describing the investigations and defining the ground water contamination problem at the landfill site. The design of the monitoring system would be based on this report. The methods and purposes for such a monitoring program are outlined in this manual.

1.7.5.1 Selection of the Monitoring Sites. Using data from the previous steps, all potential monitoring sites are ranked in order of importance. High priority sites would include currently developed aquifers and aquifers with good development potential and discharge areas (such as marshland which could be damaged by the anticipated leachate discharges). Monitoring sites should be selected to provide early warning, allowing corrective action to be taken. Ideally, monitoring should indicate the size and type of abatement program necessary in order to correct a problem once it has been detected. At the very least, the monitoring program should insure that a health hazard does not arise.

1.7.5.2 Determination of Monitoring Objectives. After selection of the monitoring sites has been made, the specific objectives of the monitoring program should be determined. In addition to the basic monitoring purposes described earlier in this chapter, specific technical objectives might include:

- defining the rate of leachate plume movement;
- monitoring the concentration of a specific contaminant(s);
- early warning of an unexpected change in direction or size of the leachate plume;

- or unexpected interaquifer movement of the plume to a previously unpolluted aquifer.

Once the monitoring objectives have been determined, the data requirements must be defined to satisfy these objectives. Data requirements would include:

- specific chemical constituents to be included in analyses of water samples;
- physical measurements to be made at the site;
- and the frequency of sampling or measurement.

For example, if an objective of a monitoring program is to insure that leachate does not migrate into a particular aquifer, monthly measurements of the specific conductance of water samples from that aquifer might be selected to provide information which will protect the aquifer at a minimal cost. A discussion of monitoring data requirements is presented in Chapter 4.

1.7.5.3 Establishing the Monitoring Methods and Procedures Necessary to Accomplish Objectives. To accomplish the specific monitoring program objectives, certain monitoring devices would be required. For example: At a specific point, a single well screened over a small section of an aquifer may suffice. However, a cluster of several wells, screened over different portions of the aquifer or in separate aquifers, may be required. Wells of a particular material may be necessary to avoid interference with leachate sample chemistry, or devices other than wells might be required. A discussion of monitoring and sampling techniques is presented in Chapter 3.

To insure uniform results, a detailed sampling or measuring procedure should be established; and, if possible, one person should be responsible for sampling. This would be especially important for complex procedures, but less so for simpler tasks (such as conductance measurements). Proper handling and storage of water samples is extremely important. For example, when nitrogen analyses are to be made, chilling or acidification of the sample is required; when metals are to be tested, acidification with nitric acid is necessary. A discussion of sample preservation is given in Chapter 6 and analytical methods in Chapter 7. The cost of monitoring will vary widely and will depend upon the sampling procedures and analyses used. Therefore, the program design should be properly operable within the available budget.

Budget allotments should be made for proper data reduction, recordkeeping, and periodic data review. Data records should be in three forms; the original data as gathered and accompanied with explanatory notes, continuous tabular form, and continuous graph form. Plotting data on near-uniform grids permits relative values and trends to be easily recognized. Proper procedures for handling monitoring data mandate the periodic review of all data by a qualified scientist followed by a written summary and the distribution and review of the summary by all involved parties.

1.7.5.4 Establishing a Management Program. Conditions under which abatement procedures, other corrective measures, or additional monitoring steps that will be taken should be outlined at the outset of monitoring. Such conditions might include; constituent limits, physical parameter limits, or trend shifts. In the event the established conditions are exceeded, possible steps which might be taken should be determined. Responsibility should be assigned for all phases of the monitoring and potential abatement programs.

1.8 EXAMPLES OF LANDFILL CONTAMINATION PROBLEMS

The following two scenarios depicting fictitious landfill investigations leading to ground water monitoring programs are somewhat different. The first begins with a landfill and defines the pollution problem; whereas the second starts with a problem and searches for its probable cause. The first scenario follows closely the preceding step outline; however, the second is an example of a problem requiring a somewhat different approach. The scenarios are intended as illustrative examples and as such are necessarily simplified, i.e., some points included in the step outline have been omitted. Since no attempt has been made to include all possibilities, other approaches and conclusions may be equally valid. These examples should prove beneficial if expanded upon using the factual information presented in other chapters of this manual.

1.8.1 Senario 1 -- A Landfill Contamination Study. A large county-maintained landfill is discovered to be allowing leachate to flow into and contaminate an adjacent river violating the 1899 Harbors and Rivers Act. As a result of a federal lawsuit, a court order is issued mandating county officials to take the necessary steps to remedy this condition. A team having ground water and engineering expertise is formed to investigate leachate conditions at the landfill site; determine if leachate is actually discharging into the river; and if so, what steps should be taken to rectify this problem.

The team assigned to this project makes a visit to the landfill for a preliminary inspection tour with the landfill operator. During this tour, it is learned that the landfill receives approximately 1,000 tons of solid waste per day. Approximately 90% is municipal waste; the remaining 10% is of industrial origin. The solid waste receives no pretreatment. However, after landfilling, it is spread into thin layers by a bulldozer and compacted by a specially designed landfill compactor. The layers of compacted solid waste are covered daily by sandy cover material. Small amounts of industrial chemical waste are accepted at the landfill; but it is not separated from the other solid waste. The rate and method of landfilling, the type of cover material used, and local precipitation rates indicate that all the solid waste has reached field capacity.

The landfill is 16.2 hectares (40 acres) in size, approximately 18.3 meters (60 feet) thick, and has a relatively flat top surface. Directly north of the landfill is a hill with an elevation of approximately 24.4 meters (80 feet). South of the landfill is a tidal marsh separating the landfill from the river. The distance between the landfill and the river is approximately 305 meters (1,000 feet). The landfill is not lined or covered with impermeable materials nor does it utilize any other leachate prevention techniques.

In the opinion of the hydrogeologist, the topography of the site indicates that ground water flow is from north to south. The hill and landfill act as recharge areas and the marsh and river as discharge areas. The nearest water-supply well serves an individual residence and is located approximately one-half mile north of the landfill. No other wells or borings are known in the landfill vicinity.

The landfill has been in existence for approximately 12 years. There has been no special site preparation prior to landfilling; solid waste has been dumped into the edge of the marsh. At present, there is little or no vegetation on most of the landfill surface, and erosion channels on the steeper slopes are apparent. Small leachate seeps are evident along most of the toe of the landfill. These flow directly out into the marsh and form leachate pools which are periodically flushed out into the river during periods of heavy rainfall. A sketch map showing important features of the landfill site is prepared by the study team during the field inspection (Figure 1).

In a discussion with the landfill operator, the hydrogeologist learns that the county is considering the construction of a berm, or dike, around the southern toe of the landfill to prevent leachate from migrating into the marsh area. County officials feel that leachate can be trapped behind such a berm and pumped to an evaporation pit or back to the top of the landfill for recirculation. The study team is asked to evaluate the effectiveness of this scheme.

Further observations by the study team reveal the fact that the flat, highly permeable top surface of the landfill would allow a large percentage of precipitation to percolate into the solid waste. In addition, surface runoff from the hilly area to the north is free to flow onto the top surface of the landfill and infiltrate into the solid waste. The volume of leachate likely to be generated from these two recharge sources would be considerably greater than the volume discharged by the surface seeps. Thus, a considerable volume of leachate must be moving with the ground-water system beneath the landfill and discharging into the marsh or

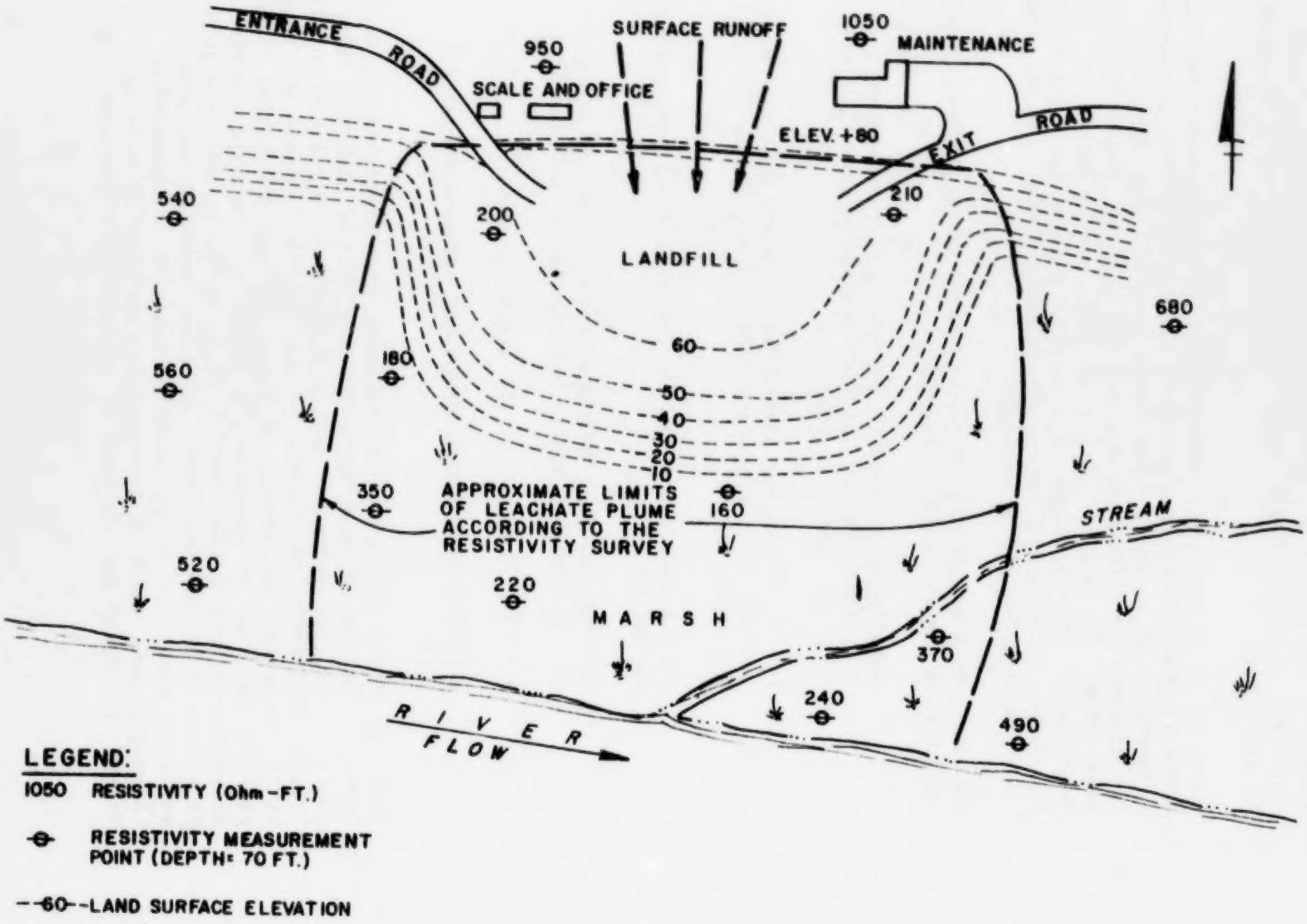

ENTRANCE ROAD
950
SCALE AND OFFICE
SURFACE RUNOFF
1050
MAINTENANCE
EXIT
ROAD
ELEV. +80
540
200
LANDFILL
210
680
560
180
60
50
40
30
20
10
350
APPROXIMATE LIMITS OF LEACHATE PLUME ACCORDING TO THE RESISTIVITY SURVEY
160
STREAM
520
220
MARSH
370
240
490
RIVER
FLOW
LEGEND:
1050 RESISTIVITY (Ohm-FT.)
RESISTIVITY MEASUREMENT POINT (DEPTH= 70 FT.)
60 LAND SURFACE ELEVATION

FIGURE 1. SCENARIO 1. SITE CONDITIONS AND RESULTS OF RESISTIVITY SURVEY

river. If this is the case, a surface berm would do little to abate the problem. Final observations of the site inspection reveal the obvious stress on vegetation in portions of the marsh directly south of the landfill. However, there is no visible effect of discharging leachate on the river.

Based on the preliminary investigation, the study team recommends a detailed groundwater investigation to determine whether contaminated groundwater is actually discharging directly into the river. If it is, the nature of the contaminants and the rate of their discharge should be determined. Further, it is stated that in this particular case, the construction of a berm may do little to prevent leachate discharge to the river. The results of the groundwater study, however, will suggest other, more effective procedures.

The study team is advised to proceed with the groundwater investigation and obtains the following information:

- precipitation records for the past three years from a weather station located 19.4 kilometers (12 miles) from the landfill site;
- A U.S. Geological Servey Map or State Geologic Survey map showing bedrock and overburden materials in the vicinity of the site;
- soil conservation reports and maps indicating soil types in the area;
- information regarding the depth and construction of the domestic supply well north of the landfill;
- a sample from one of the leachate seeps;
- a recent aerial photograph and topographic map of the site.

Analysis of the data gathered indicates that precipitation on the landfill surface averages approximately 1016 millimeters (40 inches) per year. It is then estimated that a minimum of 50 percent of this precipitation infiltrates the surface of the landfill. Since the landfill area is 16.2 hectares (40 acres), at least 75.7 million liters (20 million gallons) per year of leachate are generated from this source. The low-permeability crystalline bedrock which underlies the site probably acts as a barrier to leachate flow. Details regarding the nature of the surficial materials at the landfill site are not available. The domestic supply well north of the landfill was drilled to a depth of 30 meters (100 feet) and screened in a high-yielding coarse sand aquifer.

Water from this well indicates no leachate contamination nor do any of the water samples taken from the river. The seep sample, however, is highly mineralized and contains contaminants typically found in municipal solid waste leachate. A base map of the landfill site is traced from the aerial photograph.

To further define the location of contaminated groundwater at the landfill site, an electrical resistivity survey is conducted. The results of this survey (Figure 1) indicate that highly mineralized groundwater is confined to an area of the marsh directly south of the landfill. Some attenuation of contaminants in the groundwater appears to be occurring in the direction of the river as indicated by the slight increase in resistivity measurements from the landfill to the river.

While the results of the resistivity survey indicate that contaminated groundwater is indeed flowing from the landfill to the river, additional geologic and water-quality data are needed to define the problem further and to suggest effective abatement procedures. To obtain this information, a well drilling contractor is hired to install a series of test borings and wells. Subsequently, five test borings are drilled on and to the north of the landfill. Two casings with well points are installed in each boring. The locations of these borings, designated A through E for the deep wells and A' to E' for the shallow wells, are shown on Figure 2. The drilling rig cannot be operated in the marsh area; therefore, ten additional test wells are installed in this area by hand. The locations of these wells, designated 1 through 10, are also shown on Figure 2.

Table 1 gives information concerning the construction details of the wells and the elevations, temperature, and specific conductance of groundwater for all the wells installed. Based on these data, a water table contour map and geologic cross section are drawn (Figure 2 and 3 respectively). Also shown on Figure 2 is the groundwater head at each well point. As groundwater follows the flowlines from areas of higher head to areas of lower head, examinaton of the figures indicates that highly contaminated groundwater from the landfill is flowing downward into the deeper sediments beneath the landfill, flows upward, and discharges directly into the river. This analysis is supported by the specific conductances and temperature data.

While some attenuation of contaminants is occurring along the flow path, the attenuation is by no means complete. Detailed chemical analyses of water samples from all the test wells (not given here) confirms this opinion. Contaminated ground water from portions of the landfill is discharging directly into the marsh (Figure 3) and probably is responsible for the observed vegetation stress. Figure 3 also shows contaminated water discharging directly to the river. The dilution is so great, however, that this source of contamination is not detectable in river-water samples.

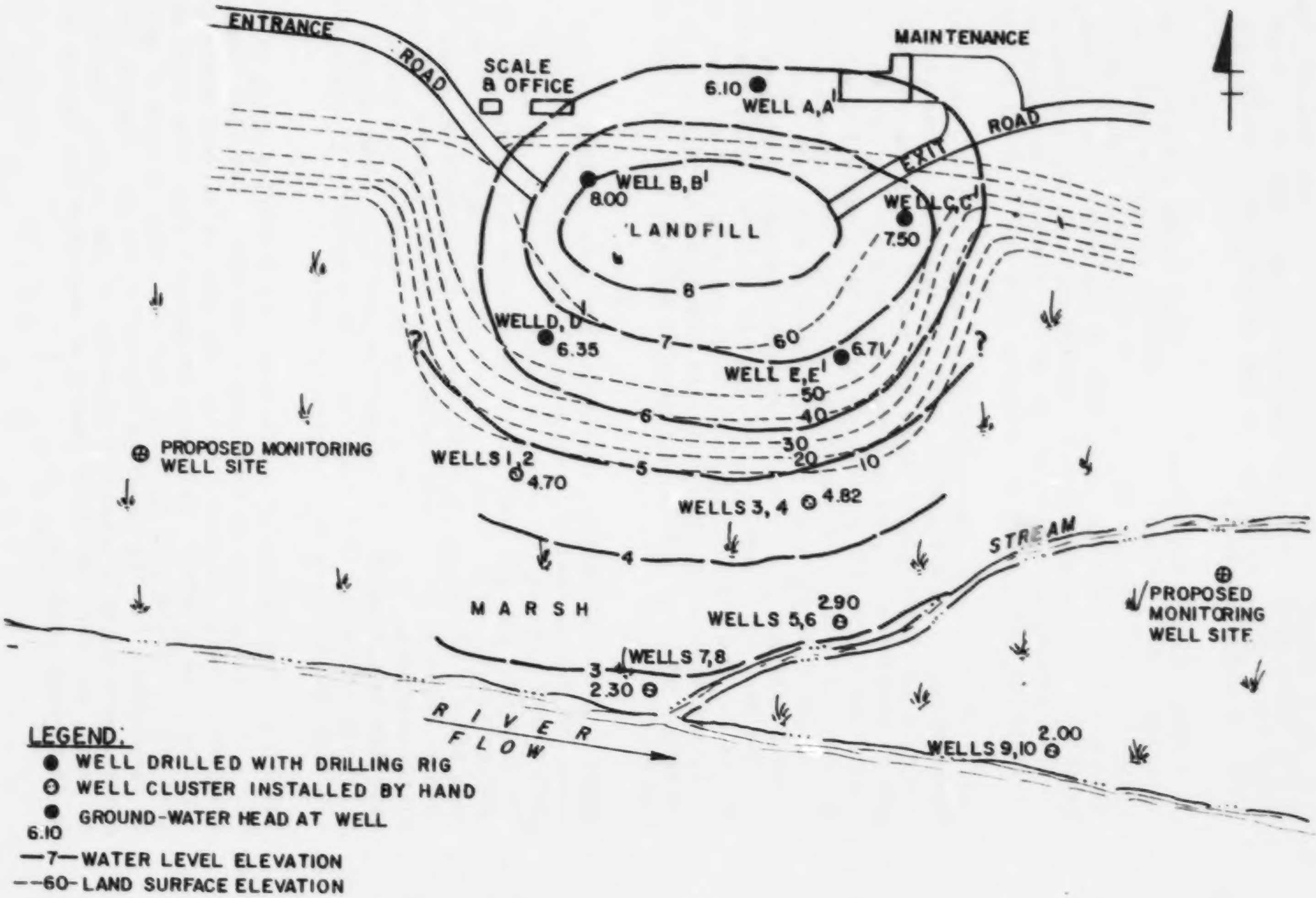

ENTRANCE ROAD
SCALE & OFFICE
MAINTENANCE
EXIT ROAD
6.10 WELL A,A'
WELL B,B' 8.00
LANDFILL
WELL C,C' 7.50
8
WELL D,D' 6.35
7
60
6.71 WELL E,E'
50
40
6
30
20
10
5
?
?
PROPOSED MONITORING WELL SITE
WELLS 1,2 4.70
WELLS 3,4 4.82
4
STREAM
MARSH
WELLS 5,6 2.90
PROPOSED MONITORING WELL SITE
3
WELLS 7,8 2.30
RIVER
FLOW
WELLS 9,10 2.00
LEGEND:
● WELL DRILLED WITH DRILLING RIG
ʘ WELL CLUSTER INSTALLED BY HAND
● 6.10 GROUND-WATER HEAD AT WELL
—7— WATER LEVEL ELEVATION
--60- LAND SURFACE ELEVATION

FIGURE 2. WATER TABLE AND LAND SURFACE CONTOUR MAP WITH TEST WELL LOCATIONS

TABLE 1

WELL CONSTRUCTION DETAILS, WATER LEVELS AND WATER QUALITY (PHYSICAL) - COUNTY LANDFILL SITE

(all wells equipped with 2-foot screen or well point)

WELL	WELL DIAMETER (INCHES)	WELL DEPTH (FEET)	TOP OF CASING ELEV. (MSL.)	DEPTH TO WATER (FEET)	WATER ELEV. (MSL.)	SPECIFIC CONDUCTANCE uMHOS/CM.	TEMP. °F
A	2.5	102	82.20	74.20	+8.00	210	62
A'	1.5	83	82.20	74.10	+8.10	220	62
B	2.5	90	67.80	60.15	+7.65	210	62
B'	1.5	70	67.80	59.80	+8.00	270	65
C	2.5	94	68.44	60.59	+7.85	940	72
C'	1.5	73	68.48	59.98	+8.50	1060	90
D	2.5	90	58.50	51.35	+7.85	890	71
D'	1.5	58	58.50	51.15	+7.35	1120	94
E	2.5	92	57.21	49.91	+7.30	990	75
E'	1.5	60	57.32	49.61	+7.71	1220	101
1	1.5	30	7.10	2.00	+5.10	920	82
2	1.5	10	7.10	2.40	+4.70	820	68
3	1.5	30	8.00	2.80	+5.20	1000	83
4	1.5	10	8.00	3.18	+4.82	900	67
5	1.5	35	6.60	4.20	+2.40	720	69
6	1.5	10	6.60	3.70	+2.90	690	65
7	1.5	30	6.90	3.90	+3.00	480	64
8	1.5	10	6.90	4.60	+2.30	480	63
9	1.5	30	7.10	4.00	+3.10	360	63
10	1.5	10	7.14	5.14	+2.00	420	63

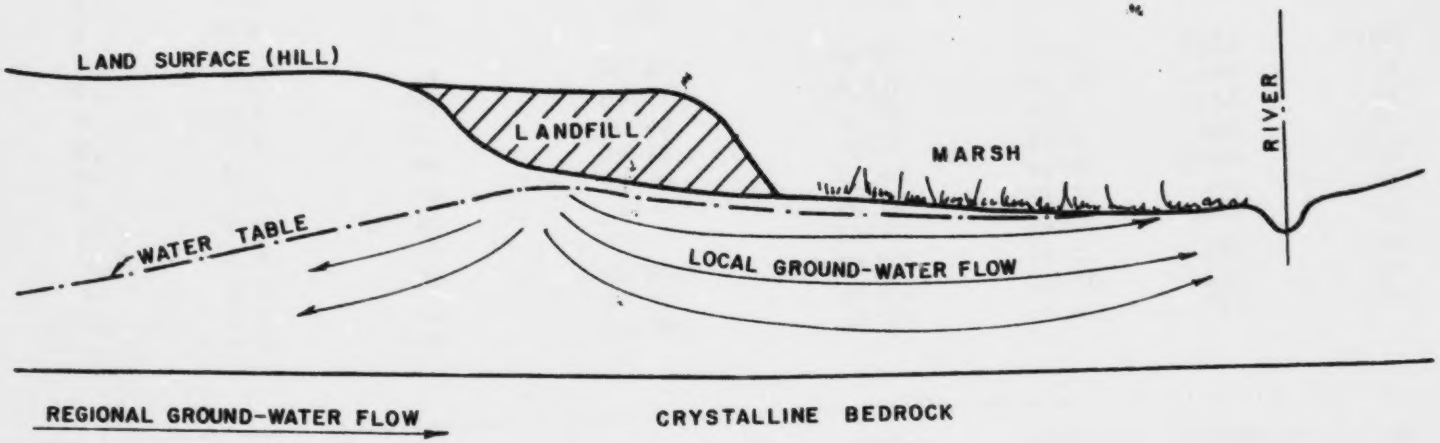

LAND SURFACE (HILL)
LANDFILL
MARSH
RIVER
WATER TABLE
LOCAL GROUND-WATER FLOW
REGIONAL GROUND-WATER FLOW
CRYSTALLINE BEDROCK

FIGURE 3. GEOLOGIC CROSS SECTION A-A'

Possible actions that might be considered with regard to this problem are as follows:

- do nothing;
- remove the landfill to a more hydrologically acceptable site;
- construct a shallow surface berm around the toe of the landfill;
- install pumping wells directly beneath the landfill to reverse the hydraulic gradient;
- install a line of interceptor wells along the toe of the landfill to restrict movement of leachate away from the landfill toward the river;
- reduce leachate generation by restricting recharge to the landfill.

The severe stress placed on the marsh by the discharging leachate renders the first possibility unacceptable. The second possibility is prohibitively expensive; and due to the deep migration of the leachate, the third possibility would do little or nothing to abate the problem. The fourth and fifth possibilities may be technically feasible; however, the low permeability of the sediments beneath the landfill would make installation difficult to accomplish. Furthermore, these two solutions would create the new problem of handling the large volume of contaminated water pumped from the wells. The final possibility appears to be the best alternative and is recommended to the county.

The recommended procedures to reduce infiltration and leachate generation are as follows:

- A more environmentally sound alternate site should be sought and prepared using the latest technology to reduce environmental impact. Phase-out operations at the existing landfill should continue to achieve an acceptable final grading plan that minimizes infiltration and leachate generation. Thereafter, the landfill should be closed as soon as the alternate site is prepared.
- A cutoff trench should be used to immediately eliminate runoff onto the landfill surface; the collected uncontaminated runoff should be drained directly into the marsh for its beneficial flushing action.
- When the landfill is closed to further dumping, the landfill surface should be regraded to eliminate any flat or depressed

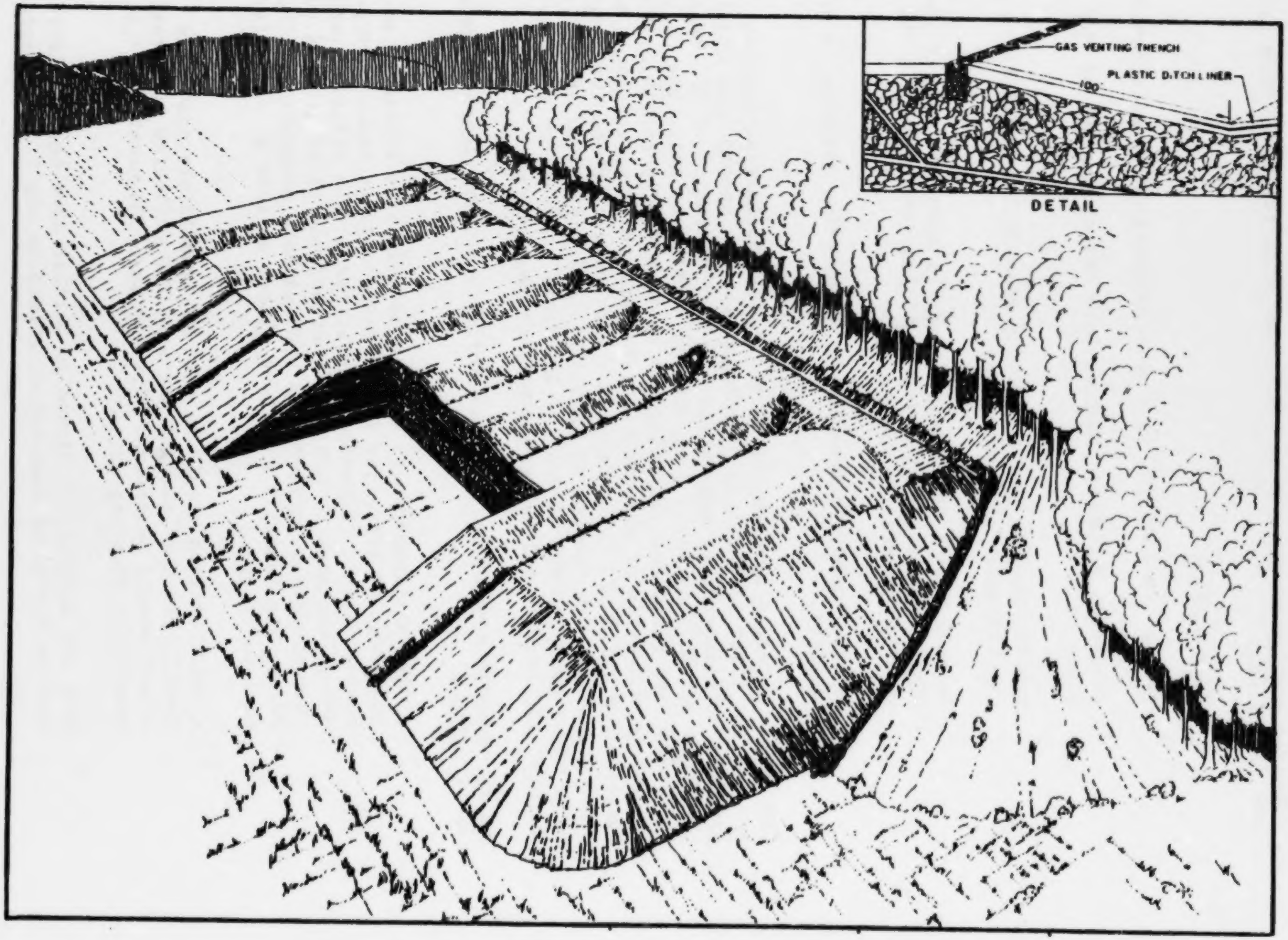

GAS VENTING TRENCH
PLASTIC DITCH LINER
100'
DETAIL

Figure 4.. Parallel Ridge Grading Plan For a Land Disposal Site

areas. A continuous grade should be created from the hill toward the marsh to encourage surface runoff. Use of a "parallel-ridge" grading plan (Figure 4) is one technique used to minimize infiltration and leachate by enhancing runoff and controlling erosion.

- The entire landfill surface should be covered with a compacted soil of low permeability. This compacted material should be covered with a layer of topsoil, and high-water use grass species planted (e.g. alfalfa).

- A series of swales and channels should be constructed on the landfill side slopes to further increase surface runoff, reduce erosion, and direct the surface runoff into the marsh beyond the toe of the landfill.

In addition to the preceeding suggestions, the study team recommended that the county institute a monitoring program to determine the effectiveness of the abatement plan. It is recommended that the monitoring data collection begin as soon as possible to obtain antecedent information prior to making the abatement improvements on the completed landfill.

The recommended program is as follows:

- Monitoring wells of the same design as Well No. 2 (well cluster) should be installed at the two locations marked ⊗ on Figure 2. These should be used as monitoring wells in addition to the 15 existing test wells.

- The water level in each well should be measured once a month.

- A yearly water sample should be taken from each well and an expanded chemical analysis conducted of each sample taken.

- If any well shows a marked change in specific conductance, an immediate analysis should be done of a water sample taken from that well for selected indicator analyses.

- A rain gauge should be installed on the landfill surface and the monthly precipitation recorded.

- All data should be reduced to both tabular and graph form.

- All data should be reviewed annually; if necessary, the monitoring program should be adjusted as suggested by the data analyses.

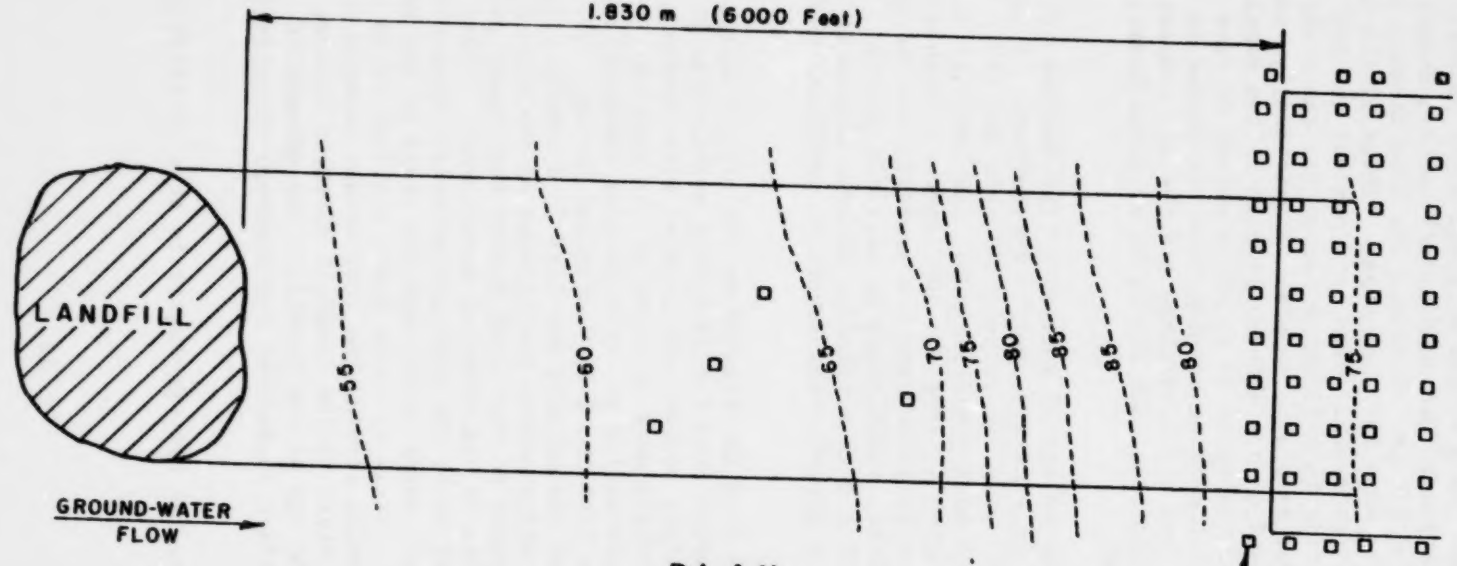
1.8.2 Scenario 2 -- A Ground Water Contamination Problem - Over a period of two months, a growing number of complaints have been registered by residents of a housing development at the northern edge of a small city. Thus far, eight citizens from the development have visited the Board of Health to complain that their drinking water suddenly has become unsuitable.

The city sanitarian sends an inspector to investigate the eight complaints. He returns with the following report: in two of the houses, slightly reddish water comes from the faucets (even after running for prolonged periods). In a third house, the water is slightly gray. Although the remaining houses are not experiencing discoloration, the water does have a peculiar taste and a slight odor.

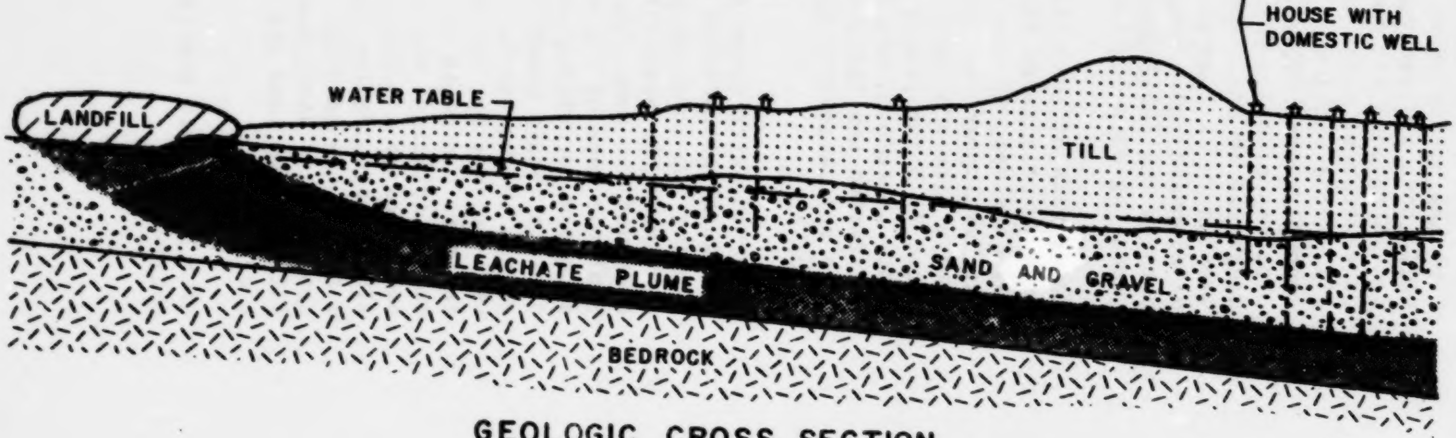
Since none of the houses has a water softener, the inspector collects a water sample directly from the kitchen sink from each house. The water samples are sent to a laboratory for analysis. Meanwhile, the sanitarian maps the location for each of the affected houses. He notes that each house is connected to the city sewer system, but each has its own water-supply well. All of the houses where the problem has occurred are in the northern half of the development and within an area a quarter-mile wide. Dozens of other houses are interspersed with the ones inspected, each having the same type of water supply and waste-disposal system.

Apparently, there are only two possible causes for the water-quality problems. The more likely of the two is that the 10-year old sewer system may have suddenly developed several large leaks causing the raw sewage to seep into the ground and contaminate the wells. The more remote of the potential causes would be the 18.2 hectares (45-acre) county landfill located more than 1.6 kilometers (1 mile) north of the nearest affected house. When considering the topography of the area, the landfill as a cause seems even more unlikely. North of the development the terrain rises gently for several hundred yards and then is broken by a steep, elongated hill blocking a view of the landfill from the city. Beyond the hill, the ground slopes gently downward for more than .8 kilometers (half a mile) to the edge of the landfill, located in an old gravel quarry (Figure 5).

The sanitarian concludes that for contaminated water from the landfill to reach the northern development it would have to travel in an uphill direction for over a mile. If this were the case, the question arises as to why the other houses were not affected sooner since both the development and the landfill have existed for ten years. Furthermore, why were only a few houses in the development affected? If the landfill were the cause, why were the four houses located north of the hill between the development and the landfill not affected? In an attempt to define the problem, the sanitarian sends his inspector to obtain water samples from several additional houses in the development where no problem has been reported. Samples are also taken from two of the four houses between the landfill and the development.


1.830 m (6000 Feet)
LANDFILL
55
60
65
70
75
80
85
85
80
75
GROUND-WATER FLOW

PLAN


HOUSE WITH DOMESTIC WELL
WATER TABLE
LANDFILL
TILL
LEACHATE PLUME
SAND AND GRAVEL
BEDROCK

GEOLOGIC CROSS SECTION

FIGURE 5. SCENARIO 2. GEOLOGIC CROSS SECTION AND PLAN

In this case, the results of the water analyses did little to indicate the source of the problem. Each of the original eight samples (taken from the houses where the owners had complained) contain constituents well above the recommended limits. However, the constituents in highest concentrations are not the same for each house. Three of the houses have water supplies with abnormally high iron content and low pH. In all of the samples, chloride is well above normal for the area; but the concentrations differ from sample to sample. Significantly, concentrations of calcium and sodium are abnormally high in two samples and manganese in one. Ammonia is found to be above normal in five of the samples. The analyses of the three samples from the houses in the development whose owners had not complained and the two samples from outside the development indicate that the wells at these locations are producing high-quality water.

The levels of chloride and metals in several of the samples are too high to have originated from the sanitary sewer. Furthermore, if large leaks had developed in the sewer line, it is unlikely that the other houses in the development would have high-quality water. The landfill is more than a mile away and is downhill from the development. High-quality water is being pumped from wells between the landfill and the development; therefore, the landfill still seems an unlikely source. The sanitarian now believes that some completely unknown source is responsible and decides to hire a ground water expert to determine what it might be.

A ground water consulting firm is retained by the city; presented with the analyses of water samples from the 13 houses accompanied by a map showing the location of those houses; and charged with locating the source of the apparent contaminants in eight of the samples.
Assigned the task, the hydrogeologist first obtains topographic and geologic maps of the area from the U.S. Geological Survey. He contacts a company providing aerial photography services in a nearby city and is able to obtain black and white aerial photographs of the city and the region to the north. A visit to the local Health Department provides well records for the houses in the affected development. These records indicate the depth of each well, the geologic materials penetrated during drilling, the static water level, and the yield of the well as estimated by the driller. Calls to three local drilling firms produce similar records for the wells serving the four houses between the development and the landfill. A visit to the landfill site and discussions with the operator discloses the age of the landfill, the methods of landfilling used, and the surface conditions and drainage characteristics of the landfill.

With these data available, the hydrogeologist is able to establish the following:

- The houses in the development and the four houses north of the development are resting on a layer of glacial till 3 to 9 meters (10 to 30 feet) thick.

- Beneath this till layer is an extensive sand and gravel aquifer, estimated to be approximately 15 to 30 meters (50 to 100 feet) thick. Underlying this aquifer is crystalline bedrock.

- The general direction of ground water flow is from the mountainous area 16 kilometers (10 miles) north of the city.

- The gradient of the water table in the vicintiy of the development is low, but the permeability of the aquifer is relatively high. The rate of ground water flow in the area is about .6 meters (2 feet) per day.

- With the exception of the eight contaminated wells, the wells belonging to the houses in the development are screened near the top of the sand and gravel aquifer in an interval of about 15 to 18 meters (50 to 60 feet) below land surface.

- With corrections for differences in elevation, the four wells belonging to the houses north of the development are screened at approximately the same depth in the aquifer as the majority of the development houses.

- The eight contaminated wells in the development are screened substantially deeper than the other development wells. In four of these, a 3-meter (10-foot) thick clay lens was penetrated at the normal screening depth of 12 to 18 meters (40 to 60 feet); and the wells were drilled an additional 6 meters (20 feet) into the sand and gravel beneath the clay. The remaining four wells were drilled at a later date by a different drilling firm and were inexplicably deeper.

- The landfill, located 1,830 meters (6,000 feet) upgradient of the development, is situated in an abandoned gravel pit and most likely directly connected to the aquifer serving the development.

- The landfill is roughly circular, covering an area of approximately 18.2 hectares (45 acres) and is about 488 meters (1,600 feet) in diameter.

- The contaminants of high concentrations found in the eight wells in the development are characteristic of typical municipal landfill leachate.

- The landfill and the sewer system are the only significantly large sources of contamination located in the immediate vicinity of the development or upgradient of the development as far as the mountains 16 kilometers (10 miles) north.

- The landfill is 10 years old and has a broad, flat upper surface. The deposited solid waste is covered daily with sand taken from an unfilled portion of the old gravel bank.

- Rainfall in the area averages 1,010 millimeters (40 inches) per year.

Based on these findings, the hydrogeologist concludes that it not only is possible but very probable that the landfill is the source of the contamination found in the eight wells. Since the deeper wells had become contaminated rather than the shallow ones, the sewer was no longer considered the source of contamination.

Using the available geologic and hydrologic data, a cross section of the area (including the landfill and the development) illustrates how only the deeper wells could become contaminated (Figure 5). Since the landfill most likely rests directly on top of the aquifer, leachate generated in the landfill would flow into and move with the natural ground water. From other landfill investigations, however, it is known that leachate can flow as a distinct plume with relatively little dispersal in the ground water system. Furthermore, this plume may tend to sink toward the bottom of the aquifer as it moves. The plume might be of sufficient thickness to enter the deeper wells; yet, it is still possible for it to flow underneath the shallower ones (Figure 5). As previously mentioned, both the landfill and the development have existed for ten years. The reason for the 10-year delay in appearance of contamination is the estimated flow rate of the ground water. If the assumption is made that the leachate began to move into the aquifer during the first year of landfilling, the approximate time lapsed before reaching the vicinity of the first well would have been 3,000 days. The distance from the landfill to the well is 1,830 meters (6,000 feet), indicating a ground water velocity of .6 meters (2 feet) per day and correlating to established estimates of normal ground water flow. The contamination then traveled 122 meters (400 feet) from the first well affected to the last well affected in only 60 days. Normally, this would have taken 200 days. This inconsistency can be explained by the fact that the velocity of the ground water changes as it enters the cone of influence created by a large number of pumping wells in a developed area.

The width of the affected area, .4 kilometers (one-quarter mile) is approximately equal to the width of the landfill. It is possible for a leachate plume to migrate without substantial dispersion and remain at approximately its original width for substantial distances. Thus, the plume should be at least 488-meters (1,600-feet) wide (probably somewhat wider) as it reaches the development.

Along with the data and findings, the ground water consultants include the following recommendations in their report to the city:

- Immediately, the owner of the contaminated wells should be advised to obtain their drinking water from other sources.
- Water samples should be collected from all the unsampled wells in the subdivision and analyzed for concentrations of chloride, calcium, and iron. If any abnormalities exist, the owners of those wells should be advised not to drink the water.
- An immediate investigation should be instituted to positively establish the landfill as the source of the problem and to define the actual extent and rate of movement of the contaminants. Also, detailed water analyses should be performed to determine what potential health hazards exist.
- When the problem has been defined, effective abatement procedures should be established. The various possible procedures should be evaluated and a determination made as to the most effective.

The city Department of Health assumes the responsibility for the enforcement of the first two recommendations. The consulting team with ground water and engineering expertise is contracted to undertake the work necessary to satisfy the third and fourth recommendations.

1.8.2.1 Landfill Investigations. The first phase of the consultant's investigation, i.e., to establish the landfill as the actual cause of the problem and to define the nature and extent of the leachate plume, is undertaken as a series of tasks:

Task 1: Assemble and analyze all available background data (already done during preliminary investigations).

Task 2: Conduct a field inspection of the landfill site (already done during preliminary investigation).

Task 3: Conduct a resistivity survey in an attempt to define the depth and lateral extent of the leachate plume.

Task 4: Drill a total of 6 wells to bedrock depth to verify the results of the resistivity survey and to obtain geologic and water samples. Conduct pumping test to determine actual hydrologic characteristics of the aquifer. These wells will also be used in a continual monitoring program.

Task 5: Construct a water-balance model of the landfill to accurately determine the contributions of precipitation and underflow to the volume of leachate generation. This task would be accomplished entirely with existing data.

The results of the Phase I investigation indicate that the preliminary analysis of the situation was essentially correct. Furthermore, the leachate plume is found to contain hazardous constituents originating from industrial waste traditionally accepted at the landfill. The volume of leachate being generated by the landfill is calculated to be approximately 303,000 liters (80,000 gallons) per day from precipitation with no significant contribution from the underflow.

Based upon these results, two alternative abatement programs are presented to the city. Both programs begin with the elimination of the pollution source. The latter portion of each program deals with the handling of the existing leachate. Monitoring recommendations are included in both programs.

1.8.2.2 Abatement Program I. It is recommended that placement of solid waste for the existing county landfill be stopped as soon as an alternate disposal site can be located and prepared. The selection of a new site should be based upon geologic and hydrologic considerations so as not to create a new ground water contamination problem. Site preparation and landfilling methods should be based upon the latest technology to minimize the possibility of leachate contamination of ground or surface water.

The phase-out operations at the existing landfill should be planned to eliminate the flat, top surface and depressions; provide adequately sloped sides; and to promote precipitation runoff. The upper surface of the landfill should be covered with a minimum .6 meters (2-foot) layer of a low-permeability compacted soil to minimize infiltration. This upper layer should be covered with a layer of top soil and then seeded. A dense vegetation cover on the landfill surface should be maintained to maximize evapotranspiration.

It has been determined by aquifer tests that the existing leachate in the ground water system can be removed by a series of high-capacity pumping wells. Three 245-millimeter (10-inch) diameter wells would be installed at 122-meter (400-foot) intervals across the plume and 152 meters (500 feet) north of the development. In order to include the entire thickness of the plume in the screened zone, the wells would be drilled to bedrock and screened to a point 24.4 meters (80 feet) below land surface to rock depth. The wells would be pumped continuously at a rate of 126 liters per second-l/s (2,000 gallons per minute -gpm). This will establish a hydraulic barrier, blocking leachate flowing south from the landfill site toward the development. Also, leachate south of the barrier wells will be drawn back toward the wells by the induced reversal in gradient. When the polluted water has been removed from the aquifer beneath the development, the pumping rate of the barrier wells can be reduced to 44 l/s (700 gpm), and eight deep development wells can be returned to use--monitoring being continued for a period of time.

1.8.2.3 Abatement Program II. Landfilling should be discontinued and the existing landfill completed according to the description outlined in Program I. The eight contaminated wells should not be used for water supply, but kept intact as observation wells. Eight new wells should be drilled to a depth of 15.2 meters (50 feet) below land surface. These replacement wells would be screened above the contaminated zone at approximately the same depth as the other wells in the development. The contaminated plume should be allowed to flow along its natural course toward the river. Since there are no city supply wells or other private wells in its path, no additional effects will be apparent until the plume reaches the river. The length of the leachate flow path from the landfill to the river is approximately 4.8 kilometers (3 miles). During the course of flow from the landfill to the river, sufficient attenuation of contaminants may occur, rendering the leachate impotent. The progress of the plume should be monitored by a series of observation wells placed along its route. These monitoring wells will determine if attenuation is actually occurring at a significant rate and if the plume is altering its course. If at some time in the future it is determined that the contaminants within the plume are not being sufficiently attenuated and will be deleterious to the river, a series of barrier wells should be installed to intercept the plume prior to its reaching the river.

The city should consider the possibility of connecting the northern development to the city water supply. While this appears not to be immediately necessary, continued close monitoring of the location of the plume in the vicinity of the development may detect an enlargement of the plume and all the wells would have to be abandoned.

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1. Personal Communication. Ron Hoffer, State of Connecticut, Department of Environmental Protection to Burnell Vincent, Project Engineer, U.S. Environmental Protection Agency. 1976.

2. Office of Water Supply and Office of Solid Waste. U. S. Environmental Protection Agency. The Report to Congress: Waste Disposal Practices and Their Effects on Groundwater. January 1977

2.0 MONITORING NETWORKS

2.1 MONITORING APPROACHES

One of the basic objectives of a landfill monitoring program is to detect and evaluate potential or existing ground-water degradation caused by landfill leachate. Monitoring may be either active or passive. The former has a measurable, continuing impact on the ground-water regime, i.e. considerably altering the flow system in which the contaminant source is located. An active monitoring system might consist of one or a series of pumping wells which intercept ground water from the area that could be affected by the contaminant. Theoretically, any contaminant entering the zone of intercepted ground-water flow would eventually be detectable in the monitoring well discharge. This approach is most suited for point-source, "one-shot" contamination introduced into the ground water from such events as spills or tank leaks. In its application to sanitary landfills, this type of monitoring scheme unfortunately has several drawbacks:

- the larger (areally) the contaminant source, the greater the number of pumping wells required to intercept ground-water flow;
- disposal of the pumped water can pose a problem, especially when the water is contaminated;
- over a period of years, cumulative pumping costs and well maintenance costs may be high;
- pumping may accelerate the spread of leachate through the aquifer, and the monitoring system may eventually become a pumped withdrawal system;
- improper selection of screen depth could prevent the well from intercepting the leachate plume.

On the other hand, passive monitoring is well-suited to monitoring landfill leachate. In this approach, wells or other monitoring devices strategically located with reference to ground-water flow directions are sampled at regular intervals to determine changes in concentrations of chemical constituents in the ground water. Flow pattern disruptions are kept to a minimum. This system can be used to monitor continuous, long-term contaminant input from a source such as a landfill. To monitor the area which might be affected by the contaminant, a line of non-pumping wells is required.

The passive monitoring system will normally require a larger number of wells than the active system. However, if these wells are small diameter, sampling costs will be kept to a minimum; in the long run, this approach should cost less than a major pump installation.

2.1.1 Data Requisites for Monitoring Network Design

The basic data that should be carefully evaluated in designing a monitoring network include:

- ground-water flow direction;
- distribution of permeable and impermeable ground material;
- permeability and porosity;
- present or future effects of pumping on the flow system;
- background water quality.

Prior to commencing field investigations, the engineering/hydrogeologist study team should first contact state and federal agencies for data and publications concerning existing conditions of the landfill site and its vicinity. State environmental departments and the U.S. Geological Survey Offices are usually a valuable source of data useful to site investigation. These data may save time, effort, and expense. With this information, an engineering/hydrogeologist study team familiar with the ground-water hydrology in the area may be able to estimate conditions without actual field measurements. However, every effort should be made to perform field measurements at the site, including the installation of a series of low-cost wells, collecting geologic samples during drillings, and measuring water levels in the completed wells. Background-water quality can be determined from chemical analysis of water samples from these wells. With good statistical information, monitoring wells can be placed to most effectively detect the contaminant plume spreading from the landfill.

2.1.2 Monitoring Networks for Sanitary Landfills

In order to detect and evaluate potential or existing ground-water contamination at a landfill, a minimally acceptable monitoring well network should be implemented and consist of the following:

- one line of three wells downgradient from the landfill and situated at an angle perpendicular to ground-water flow, penetrating the entire saturated thickness of the aquifer;

- one well immediately adjacent to the downgradient edge of the filled area, screened so that it intercepts the water table.

- a well completed in an area upgradient from the landfill so that it will not be affected by potential leachate migration.

The size of the landfill, hydrogeologic environment, and budgetary restrictions are factors which will dictate the actual number of wells used. However, every effort should be made to have a minimum of five wells at each landfill and no less than one downgradient well for every 76 meters (250 ft.) of landfill frontage.

Even if wells are sited according to the background information previously described, there is a high probability that one or more of them will not intercept the plume of leachate-enriched ground water. This will be due to the heterogeneous and anisotrophic elements in aquifer material. Also, the sequence of landfilling operations has a significant effect on the shape of the leachate plume--thus, the possibility of a well not detecting leachate (Figure 6). For these reasons, if the budget allows, it is better to have too many monitoring wells rather than too few.

Once contamination is detected, additional lines of wells can be constructed farther downgradient to gauge dispersion and attenuation of the leachate thereby providing the information necessary for predicting the ultimate fate of the plume (assuming its veritcal distribution can be delineated). This approach will be a time-consuming and expensive process, the necessity for which will depend on regulatory requirements.

Provided it is properly constructed, a single well adjacent to or within the landfill can indicate whether or not leachate is reaching the ground water and will give early warning of potential aquifer degradation. Detection of leachate here should cause the landfill operator to evaluate the observations of the downgradient wells with extreme caution. Installation of additional downgradient wells at various distances from the landfill and/or implementing remedial leachate control measures may also be justified at this time. The actual course of action is dependent upon the site conditions and on federal, State, and local statutes of enforcement agencies governing ground-water contamination.

There are potential problems associated with the use of a monitoring well within the landfill which must be realized and appreciated. First, the water monitored is skimmed only from the surface of the

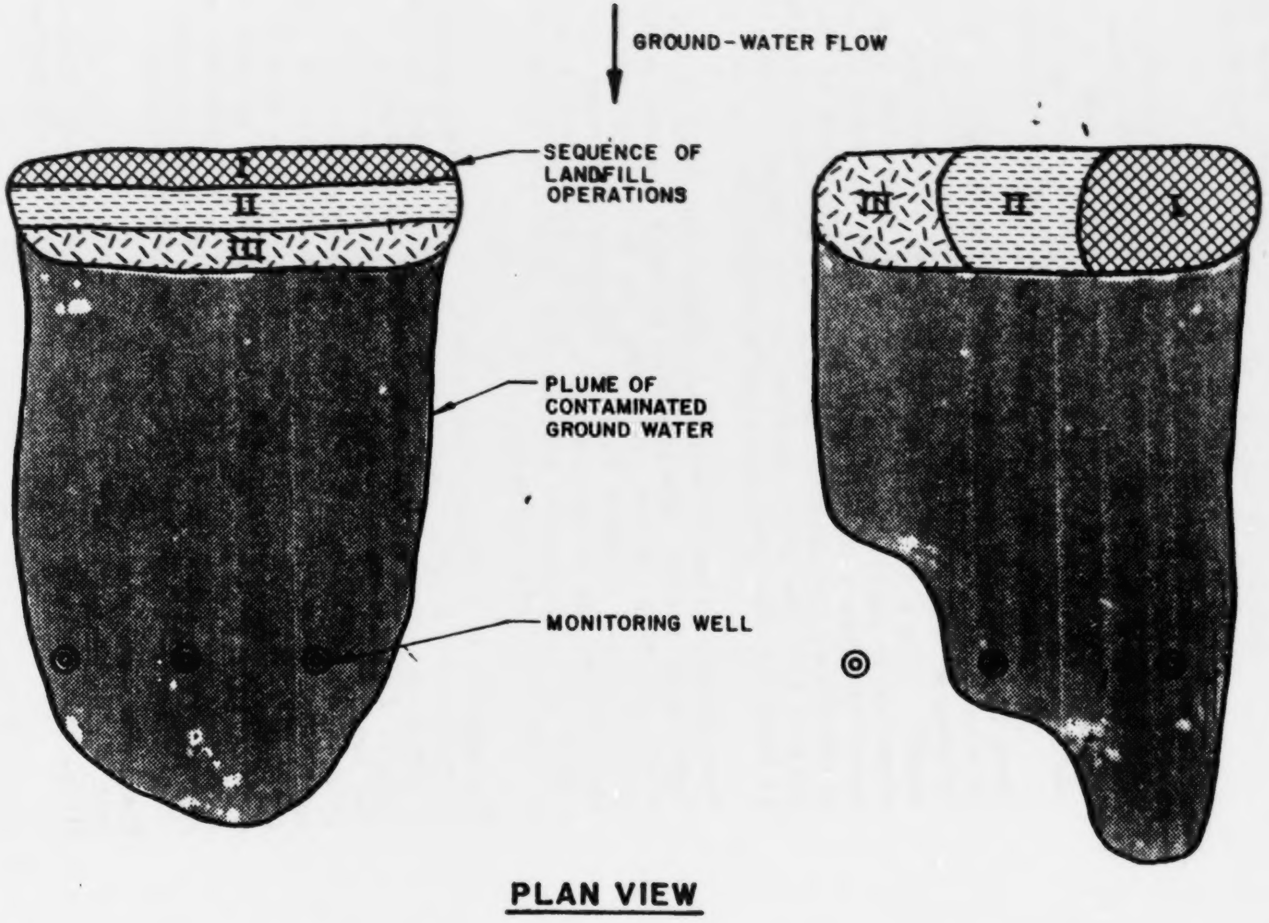

GROUND-WATER FLOW
SEQUENCE OF LANDFILL OPERATIONS
I
II
III
PLUME OF CONTAMINATED GROUND WATER
MONITORING WELL
PLAN VIEW

FIGURE 6. INFLUENCE OF LANDFILL OPERATIONS ON LEACHATE PLUMES

aquifer. If any density stratification is occurring, total reliance on this well could give an unrealistic picture of actual leachate concentration in the ground water. Secondly, elevated leachate concentrations may be found in water samples due to improper well construction. An improperly backfilled annulus can act as a conduit for downward movement of leachate, introducing it into the aquifer sooner than might have occurred naturally, (if at all). Proper construction requires the placement of an impermeable seal of either bentonite or cement grout in the annular space between the well casing and the borehole wall. Because grout can shrink and bentonite can dry and crack, their placement is not a complete guarantee of stopping downward movement of leachate. However, neglecting to place this seal during well installation is almost certain to speed and promote ground-water contamination. The value of the information obtained from this well will rarely be outweighed by the above problems, although awareness of the problems and careful well construction are of extreme importance.

The upgradient monitoring well provides water samples indicative of background water quality. This well should be sampled at regular intervals,and the analytical results used as a base line for comparison with results from the landfill and downgradient monitoring wells. The background well can also provide information on contaminants in the ground water not due to landfill leachate. When a constituent's concentration rises, an outside source is indicated. Elevated nitrates or sulfates from agricultural operations or low pH and high iron due to acid mine drainage are examples of contamination. Therefore, proper water-quality base line data are necessary for the correct interpretation of the chemical analyses of monitoring well samples, both by the operator and the regulatory agencies involved.

2.1.3 The Effects of Aquifer Characteristics on Monitoring Networks

When considering the design of the monitoring system, aquifers can be subdivided on the basis of permeability and porosity. Although the design previously described could generally be applied to all aquifers, aquifer parameters should dictate the following:

- monitoring well density, depth, construction, and drilling methods;
- probability of successful detection of the contaminant plume;
- sampling methods.

Thus, in order for the monitoring network to be effective, the basic design at a particular site will require modification according to geologic conditions. Some basic designs are presented here as guidelines--not standards; they by no means will provide answers for all of the various hydrogeologic environments found in the United States.

In an aquifer with intergranular porosity, water occurs in the void spaces between individual particles of the aquifer material. In order for the material to be considered an aquifer, these voids must be interconnected and capable of transmitting water at a useful rate. Typical aquifers are unconsolidated sands and gravels in river valleys, coastal plains, intermontane valleys, alluvial plains, and some sandstone formations. Sandstone porosity and permeability are often lower than those of unconsolidated sands due to grain cementation; cements such as iron oxide partially or completely fill the pore spaces, binding the grains together. Clay and silts, also composed of individual particles, can be highly porous; but because clay and silt are relatively impermeable, they do not readily transmit water. Typical porosity values are shown in Table 2.

Consolidated sediments and igneous and metamorphic rocks are fractured to a greater or lesser degree, depending on the intensity and frequency of the deformation and the rock type. If these fractures are interconnected and capable of transmitting water, an aquifer can form when they become filled with water. The ability of the aquifer to transmit water (permeability) depends on fracture density and interconnection; the greater the density of the fractures and the degree of interconnection, the greater the ability to transmit water. In some cases, the rocks can have both primary and secondary porosity; i.e., primary porosity (intergranular porosity) produced during sediment deposition or rock formation and secondary porosity caused by the fracturing or solution activity which follows. Sandstone is an example of both types of porosity since voids exist between the sand grains, and sandstone formations are usually fractured.

Carbonate rocks are susceptible to solution by water moving through fractures. With time, these openings are enlarged into cavities. If the cavities are connected, ground water can move very rapidly through the aquifer. In fact, carbonate aquifers can have extremely high permeability. Solution openings and sinkholes provide open pathways for leachate to reach the ground water and migrate through the formation. Carbonate rocks can have both intergranular and fracture porosity; but where solution cavities exist, ground water will move through them preferentially.

2.2 MONITORING NETWORK TYPES

2.2.1 Type I (Intergranular Porosity Aquifers)

Placement of monitoring wells in any hydrogeologic environment should be done in relationship to ground-water flow paths. This manual presents simplified diagrams to represent typical flow patterns and the positioning of monitoring wells for the type of aquifer discussed.

TABLE 2

TYPICAL INTERGRANULAR POROSITIES

Material	Porosity Per Cent
Clay	45-55
Silt	40-50
Medium to coarse mixed sand	35-40
Uniform sand	30-40
Fine to medium mixed sand	30-35
Gravel	30-40
Gravel and sand	20-35
Sandstone	10-20

Vertical and lateral flow distribution in a homogeneous, isotropic sand aquifer are shown in Figures 7 and 8. Monitoring Wells A, B, and C are the background, landfill, and downgradient wells, respectively.

The location of the background monitoring well ("A" well) is critical if there is a ground-water mound associated with the landfill. This mound, produced by increased infiltration due to landfilling, alters flow patterns in the vicinity of the landfill as depicted on Figure 7. If an "A" well is situated too close to the mound, sampling results will indicate an anomalous water-quality base line. Since the extent of flow toward the well will depend upon a variety of factors including the amount of infiltration through the landfill and aquifer characteristics, no rules of thumb can be given as to the separation between this well and the landfill proper. The best policy would be to locate the well at the most distant upgradient point of the landfill site or, if permission of the owner can be obtained, on adjacent upgradient property. Assuming there are no apparent contamination sources in the vicinity, well depth is not critical; but better base line data would be provided if the well were screened through the saturated thickness of the aquifer.

"B" wells should be constructed with great care and preferably located in the first section to be filled. Since "B" wells will detect leachate entering the ground water, monitoring the zone of aeration in hydrogeologic environments where the water table is 1.5 to 3 meters (5 - 10 feet) below the landfill is probably not necessary. However, where the unsaturated zone is 3 meters (10 feet) or greater in thickness, some monitoring device could be installed to detect downward percolation of leachate before it reaches the water table. Pressure-vacuum lysimeters can be used to trace downward movement of leachate in the unsaturated zone and will also provide data on the amount of attenuation and the likelihood of leachate reaching the water table. At a new landfill, "B" wells and any lysimeters could be installed just prior to landfilling, thus avoiding well installation through the landfill. At most existing landfills, installation of "B" wells will normally require drilling through the landfill itself.

The downgradient wells ("C" wells) should be in close proximity to the landfill in order to detect the leachate plume as soon as possible. Once leachate enters the ground water, it is difficult to control; the sooner its presence is noted, the easier it will be to initiate remedial action. "C" wells are shown screened through the entire saturated thickness of the aquifer. This procedure is recommended because the actual flow path of the leachate plume is not known unless previously defined by head relations in a number of observation wells. The flow path of leachate-enriched ground water as shown is characteristic when the landfill is located in the aquifer recharge area. If the landfill were closer to the point of discharge (in this case, the river), the plume would probably be higher in the aquifer. Since the physical behavior of contaminant bodies is not completely

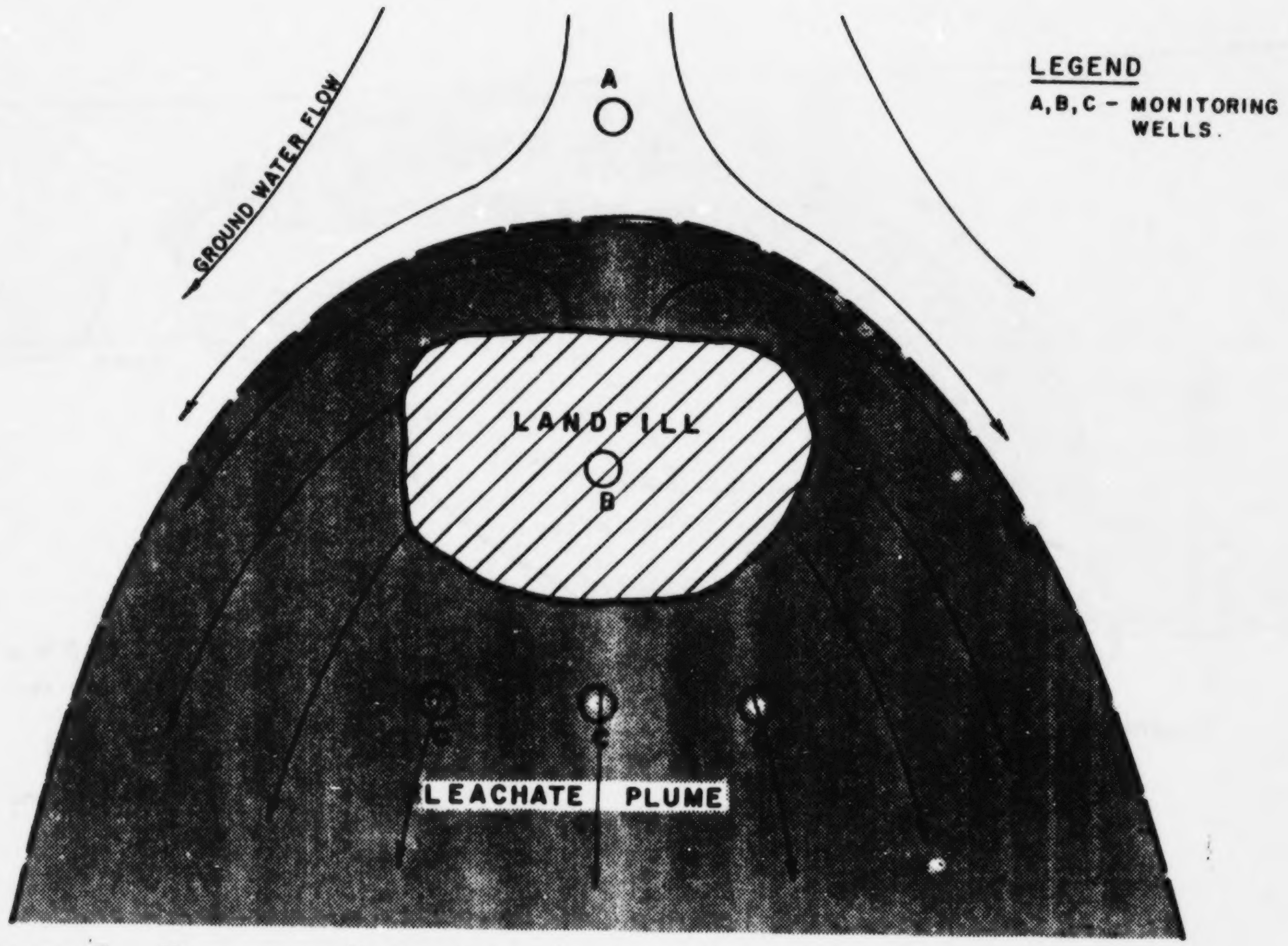

GROUND WATER FLOW
A
LANDFILL
B
C
C
LEACHATE PLUME
LEGEND
A,B,C - MONITORING WELLS.

FIGURE 7. MONITORING NETWORK FOR AQUIFERS WITH INTERGRANULAR POROSITY - AREAL FLOW PATTERNS

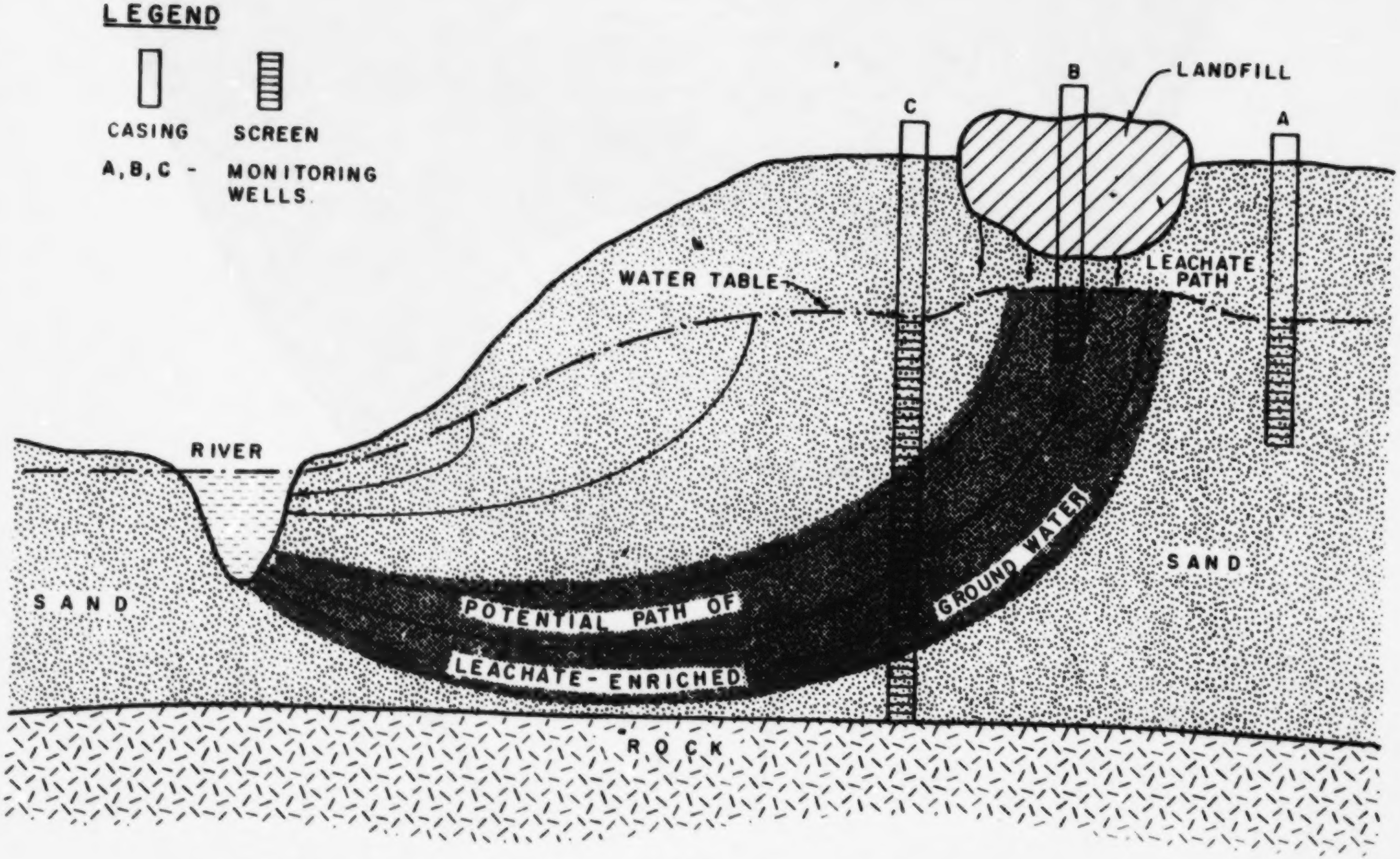

LEGEND
CASING
SCREEN
A,B,C - MONITORING WELLS
C
B
A
LANDFILL
LEACHATE PATH
WATER TABLE
RIVER
SAND
POTENTIAL PATH OF LEACHATE-ENRICHED GROUND WATER
SAND
ROCK

Figure 8. Monitoring Network for Aquifers With Intergranular Porosity - Vertical Flow Patterns

known, there is almost no way of anticipating where the plume of contaminant will be within an aquifer. Therefore, the monitoring device has to collect water over the entire saturated thickness of the aquifer; the simplest method is to screen the entire interval. This type of construction could cause complications. The primary problem is that the well can contribute to the vertical spread of contaminant by providing a conduit for ground-water movement. If the aquifer is 15 meters (50 feet) or less in thickness, this is not a major problem. Natural flow conditions would tend to distribute the leachate uniformly throughout the aquifer, especially in recharge areas. Thicker aquifers, 30 to 60 meters (100 to 200 feet), however, tend to have more pronounced shallow and deep flow systems; there is a chance the leachate plume would remain in the shallow flow system. A well screened over the entire saturated thickness provides a path from the shallow to deep flow systems. Of course, this is a gross simplification and is intended as a guideline only, since actual flow patterns will depend upon the hydrogeologic environment in which the landfill is located.

The problem of vertical spread of contaminants can be overcome by using the well-cluster technique as will be described in Chapter 3. However, it will be necessary for the landfill investigator to balance the extra cost of a well cluster against the possibility of promoting the vertical spread of contaminants.

Another problem with a well screened over the entire saturated thickness is the dilution of leachate below limits of detection when a sample is pumped from the well. This would result in a time lag between the first arrival of contaminated ground water at the monitoring well and its first detection in the sampled water. The magnitude of this lag is difficult to predict. However, as the contaminant travels in a definable plume with a small zone of diffusion to uncontaminated water, only a minimal length of time passes before the zone of diffusion moves past the well and the contaminated ground water starts to enter. If detection is still a problem, concentration or extraction techniques could then be used for key leachate tracers to determine the presence or absence of leachate in the sample. Since "C" wells are only designed to show the presence or absence of leachate and not vertical distribution once leachate is detected, a more sophisticated sampling device could be used; e.g. well clusters, sampling during drilling, or other methods discussed later in Chapter 3.

2.2.2 Type II (Fracture Porosity Aquifers)

Ground-water flow patterns are not as predictable in fractured rock aquifers as they are in aquifers with intergranular porosity. Unless there is a primary porosity, as in a sandstone aquifer, ground-water flow patterns will be controlled by the fracture pattern. Flow patterns are represented schematically in Figures 9 and 10. The

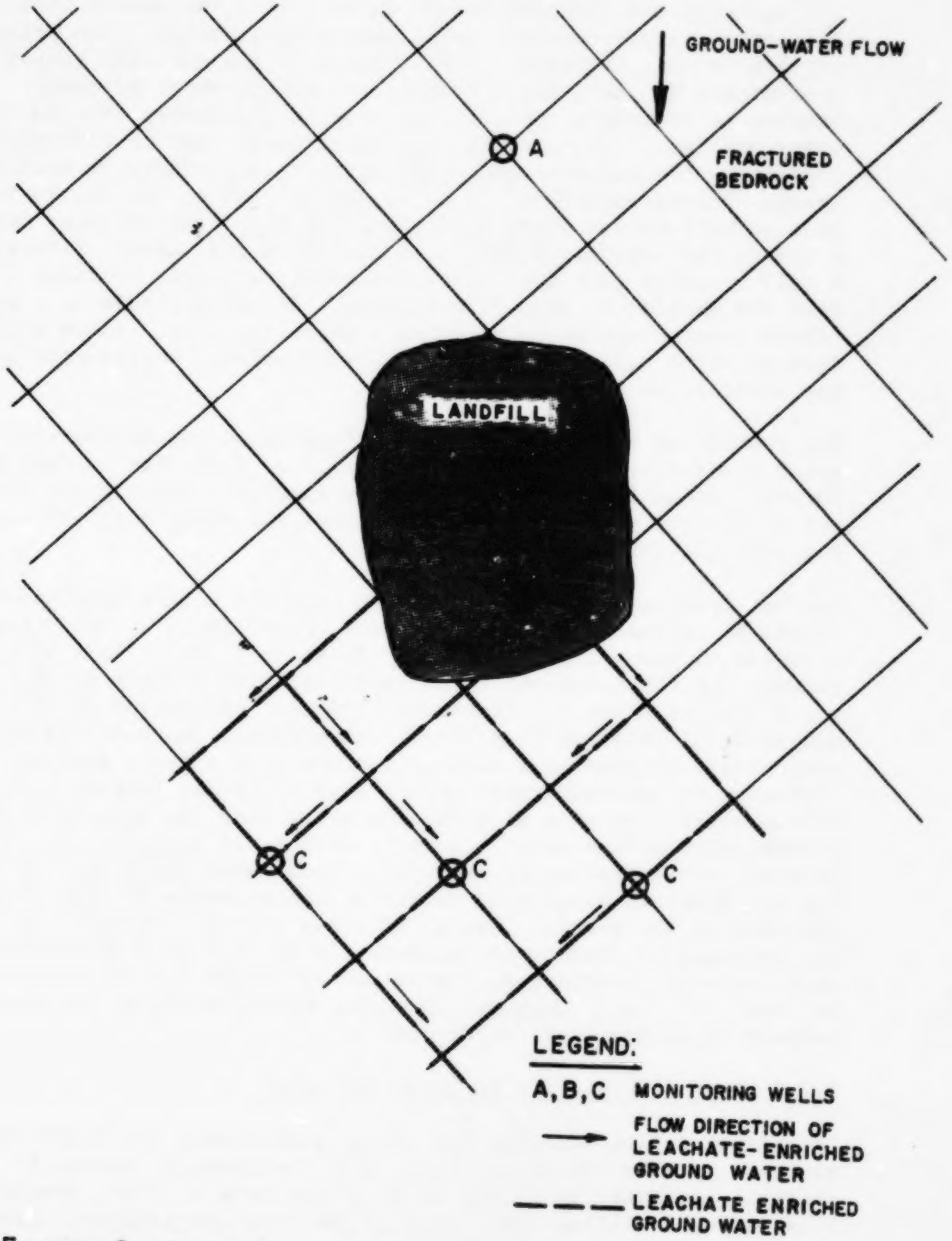

GROUND-WATER FLOW
A
FRACTURED BEDROCK
LANDFILL
C
C
C
LEGEND:
A, B, C MONITORING WELLS
FLOW DIRECTION OF LEACHATE-ENRICHED GROUND WATER
LEACHATE ENRICHED GROUND WATER

FIGURE 9. MONITORING NETWORK FOR AQUIFERS WITH FRACTURE POROSITY - AREAL FLOW PATTERNS.

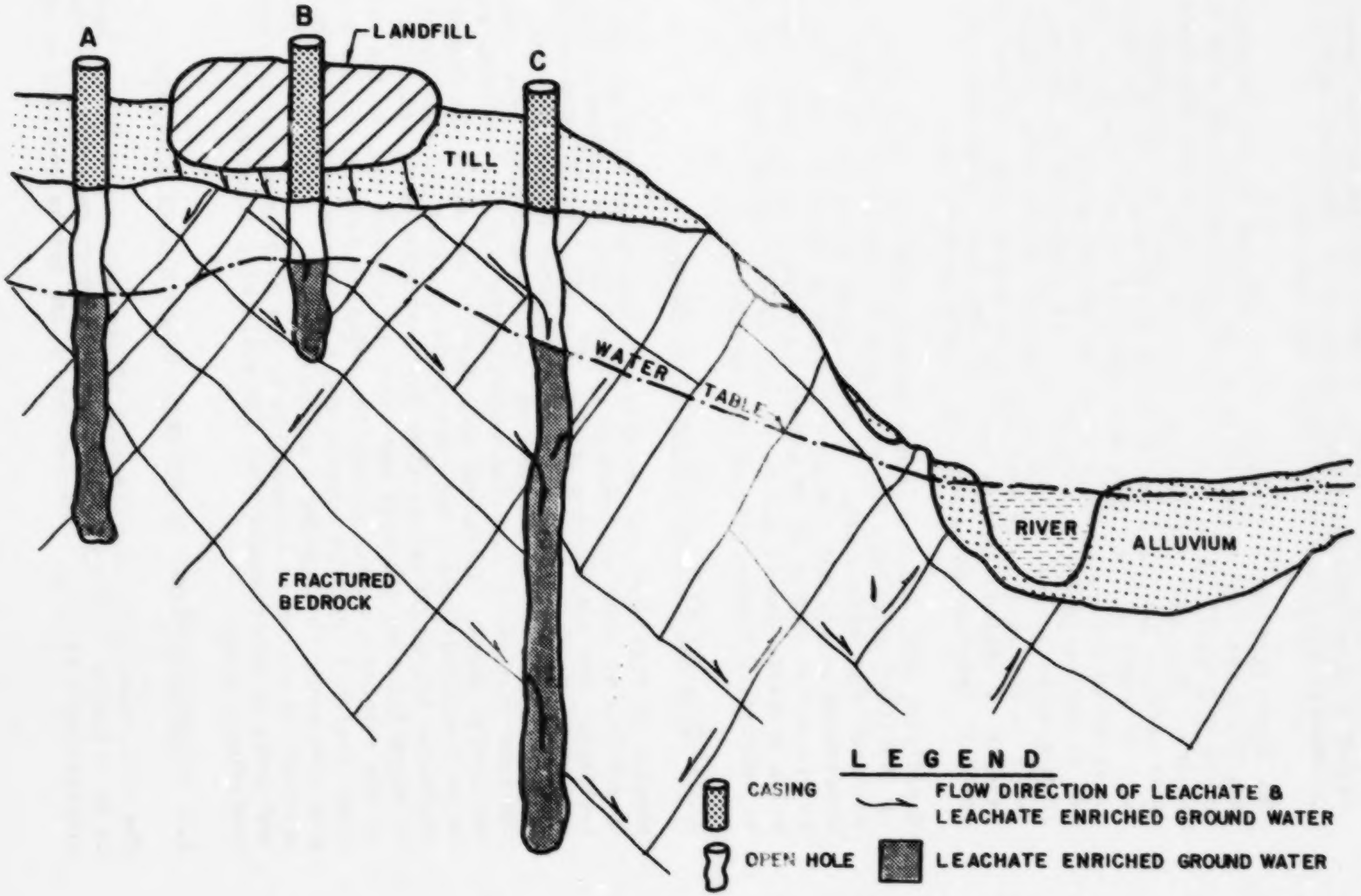

A
B
C
LANDFILL
TILL
WATER TABLE
FRACTURED BEDROCK
RIVER
ALLUVIUM
LEGEND
CASING
OPEN HOLE
FLOW DIRECTION OF LEACHATE & LEACHATE ENRICHED GROUND WATER
LEACHATE ENRICHED GROUND WATER

FIGURE 10. MONITORING NETWORK FOR AQUIFERS WITH FRACTURE POROSITY - VERTICAL FLOW PATTERNS

same configuration of "A", "B", and "C" monitoring wells may be used in this hydrogeologic regime. However, the wells are open-hole rather than screened with the exception of cased-off surficial materials to prevent them from caving into the open hole.

The major problem in fractured rock terranes is the interception of the fractures that might contain contaminated ground water with a monitoring well. A well can fail to intercept any fractures and will be dry. This will necessitate drilling another well nearby. An even more serious condition would be the interception by the monitoring well of a set of fractures not connected to the landfill, failing entirely to indicate leachate in the ground water. Without intensive and expensive geologic analysis, it is impossible to predict more than general ground-water flow direction at the site. Monitoring wells cannot be located precisely; to compensate, a higher well density is needed--perhaps one well for every 30 meters (100 feet) of landfill frontage perpendicular to ground-water flow.

Specifying well depth presents another problem. Because of the weight of overlying material, fractures either were never formed or have been squeezed shut with depth. If shut, these fractures can no longer be considered an aquifer. A general rule of thumb is that fractures tend to die out at depths of 90 meters (300 feet) or greater; monitoring wells probably should not be drilled deeper unless there is geologic or hydrologic information to the contrary.

2.2.3 Type III (Solution Porosity Aquifers)

Similar to fractured rock aquifers, ground-water flow patterns are controlled by solution openings or fractures in carbonate rock aquifers. The positioning of the "A", "B", and "C" monitoring wells in this type of flow system is shown in Figures 11 and 12. The monitoring network is the same as that for fractured rock, including the same problems: i.e. intercepting the solution cavities and well completion depth. Increased well density can solve the former; but, as in fractured rocks, there are no handy rules for the latter. Sinkholes can be 30 meters (100 feet) or more deep; while the depth at which solution cavities may exist is limited only by the thickness of the carbonate rock. Unless the solution cavities follow a well-known regional fracture system, there is no way to predict their position prior to locating a monitoring well. A trial-and-error approach to placement is mandated by the hydrogeologic environment, and there is no assurance that the wells will intercept the contaminant plume.

2.3 LEACHATE MOVEMENT IN DIFFERENT HYDROGEOLOGIC SETTINGS

The rate, direction, and distance of leachate travel from a landfill to an ultimate discharge point will be largely determined by the hydrogeologic setting. The leachate plume may be confined to the

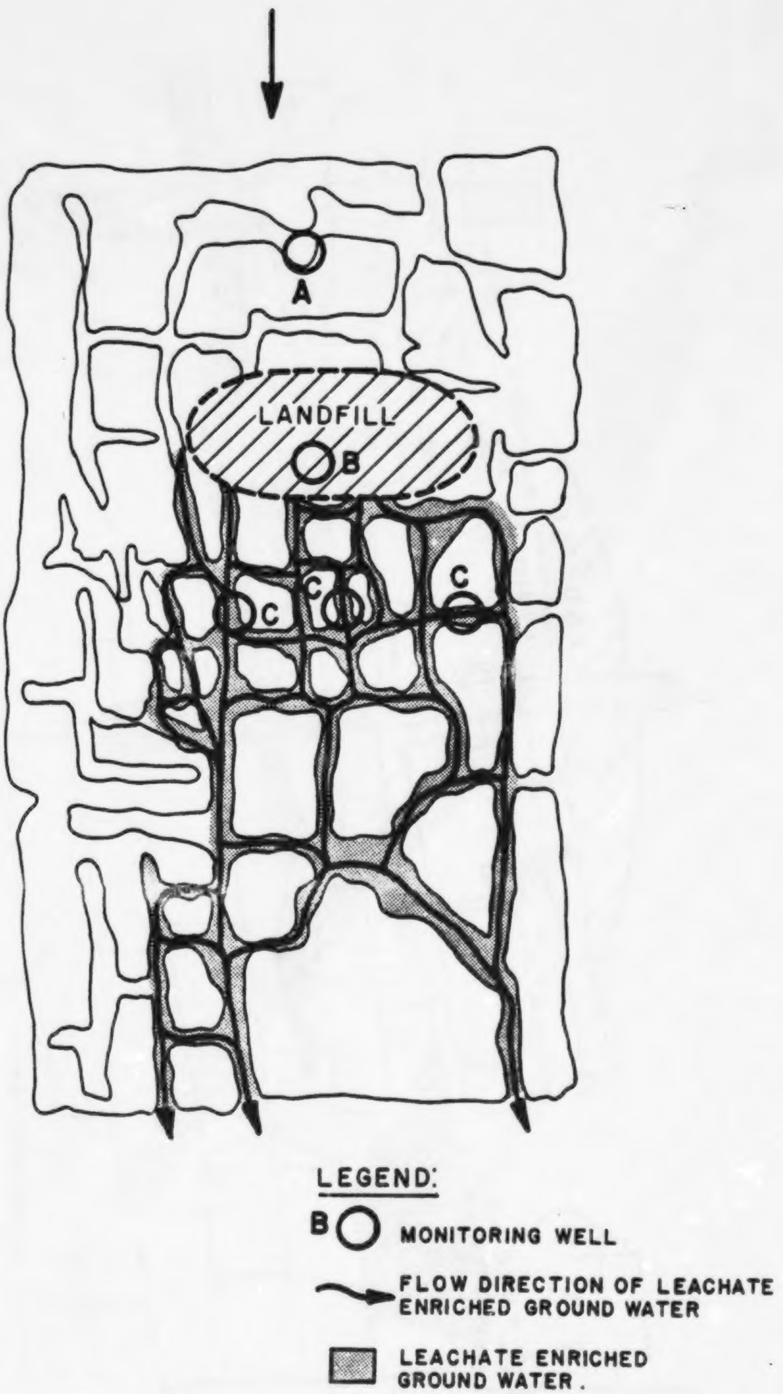

A
LANDFILL
B
C
C
C
LEGEND:
B MONITORING WELL
FLOW DIRECTION OF LEACHATE ENRICHED GROUND WATER
LEACHATE ENRICHED GROUND WATER.

FIGURE 11. MONITORING NETWORK FOR AQUIFERS WITH SOLUTION POROSITY - AREAL FLOW PATTERNS

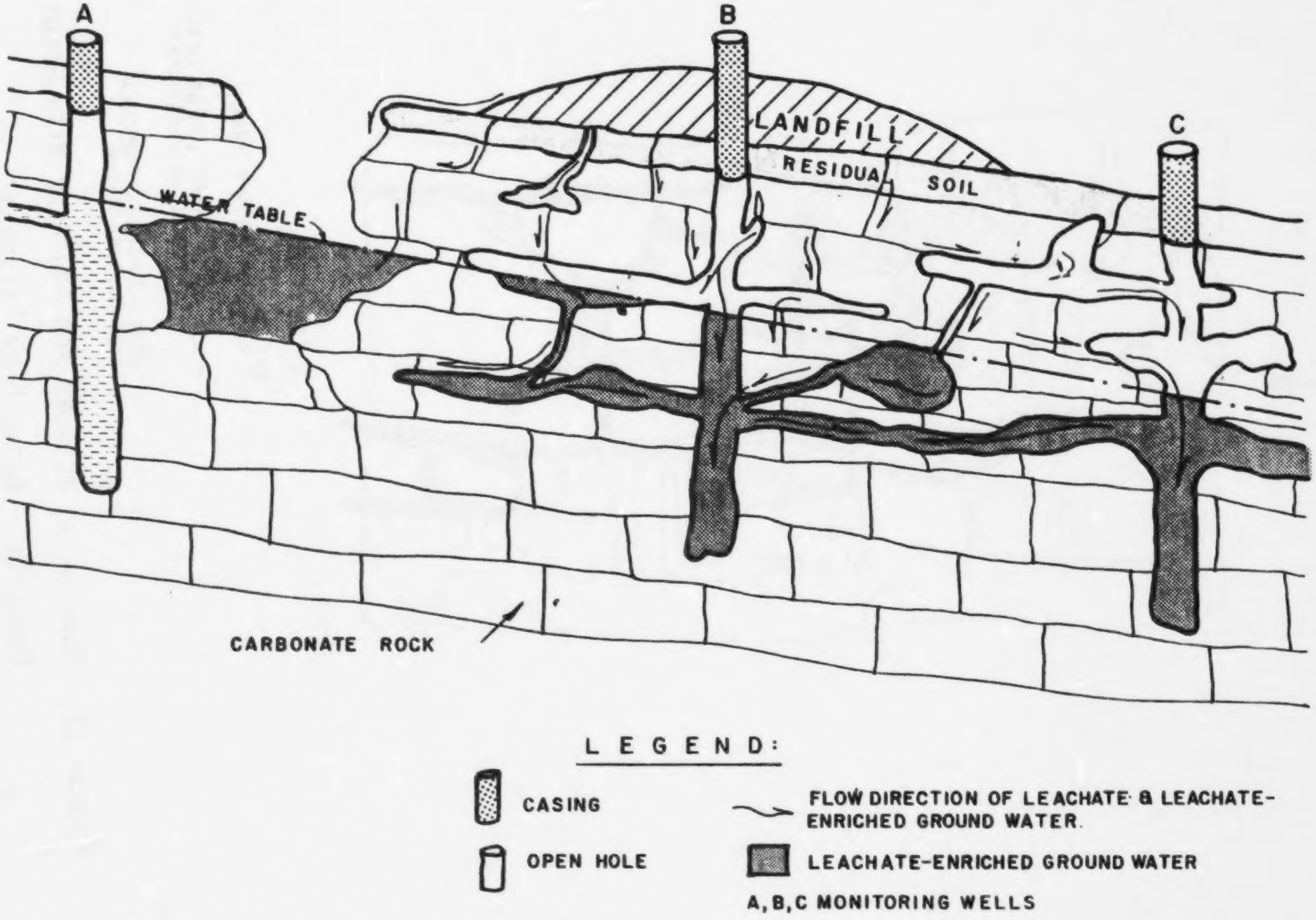

A
B
C
LANDFILL
RESIDUAL SOIL
WATER TABLE
CARBONATE ROCK
LEGEND:
CASING
OPEN HOLE
FLOW DIRECTION OF LEACHATE & LEACHATE-ENRICHED GROUND WATER.
LEACHATE-ENRICHED GROUND WATER
A, B, C MONITORING WELLS

FIGURE 12. MONITORING NETWORK FOR AQUIFERS WITH SOLUTION POROSITY-VERTICAL FLOW PATTERNS.

landfill site, or it may travel long distances; it may be divided into multiple plumes, move into different aquifers, and reverse its direction. If it is to be effective, a landfill monitoring program must account for all possible routes of leachate movement.

Figures 13 to 27 illustrate a number of hypothetical hydrogeologic landfill settings. These diagrams are schematic and are only intended to illustrate general leachate flow principles. Although both the geology and hydrology of the settings are necessarily somewhat simplified over most actual conditions, the general principles illustrated are still valid. In addition, such complicating factors as differential attenuation of contaminants by subsurface sediments, interference with leachate flow by production wells, and surface emergence of leachate have been omitted. It is obvious that if all factors influencing leachate migration from a landfill were considered, the number of possibilities would be almost limitless. For this reason, each individual landfill should be subjected to a hydrogeologic investigation prior to the establishment of a monitoring system or implementation of a pollution abatement program.

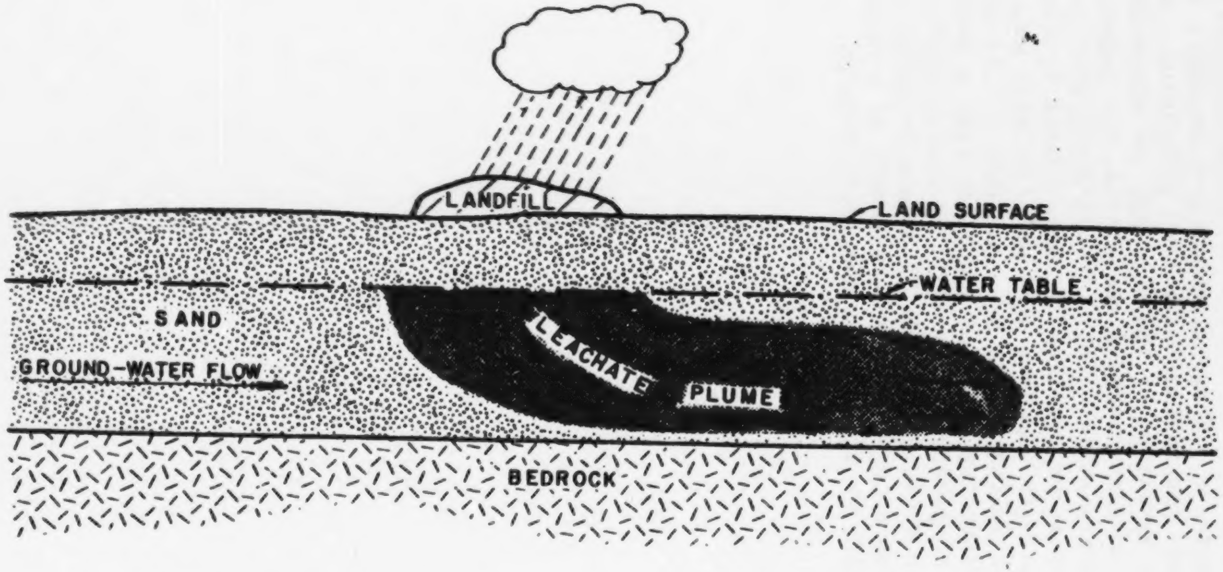

LANDFILL
LAND SURFACE
WATER TABLE
SAND
GROUND-WATER FLOW
LEACHATE
PLUME
BEDROCK

FIGURE 13. Single Aquifer With A Deep Water Table - Leachate percolates downward from the landfill to the underlying aquifer and then moves downgradient as a bulb or plume in the direction of ground-water flow. The mass of leachate may: 1) Sink to the bottom of the aquifer if of a heavier specific gravity, or 2) float at or near the top of the water-bearing unit if the leachate is predominately hydrocarbon in nature.

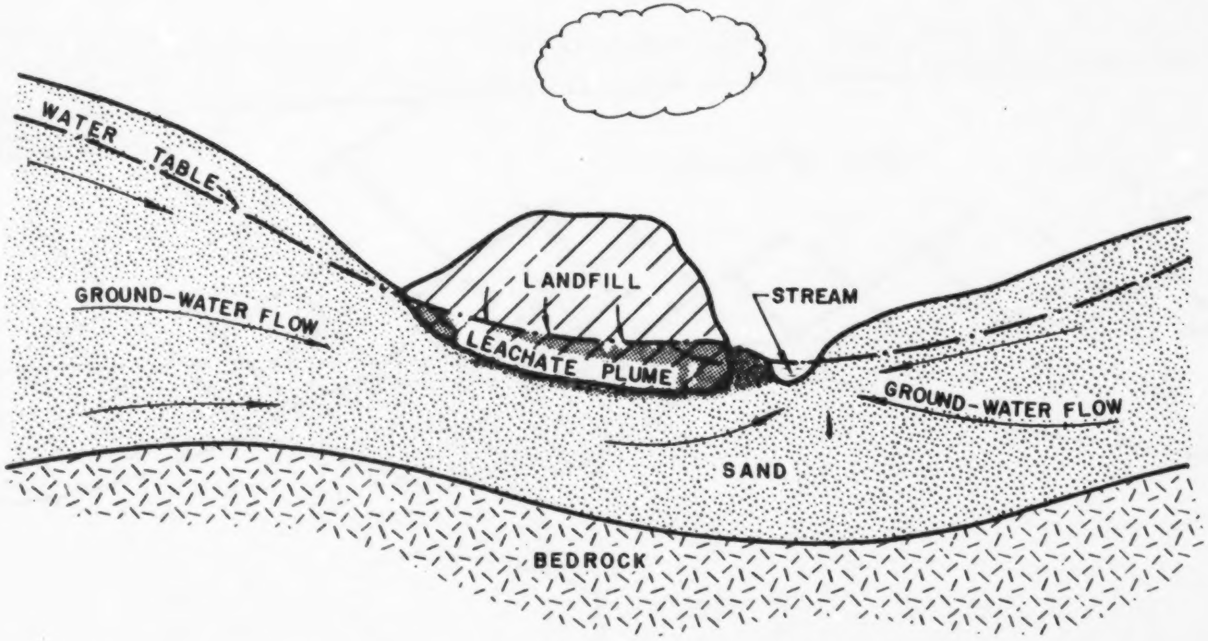

WATER TABLE
GROUND-WATER FLOW
LANDFILL
LEACHATE PLUME
STREAM
GROUND-WATER FLOW
SAND
BEDROCK

FIGURE 14. Ground-Water Discharge Areas - Landfills located within the zone of saturation are always in contact with ground water moving from topographically higher recharge areas to a stream discharge point. In such cases, leachate is transported with the ground water to the stream where it becomes diluted by surface water.

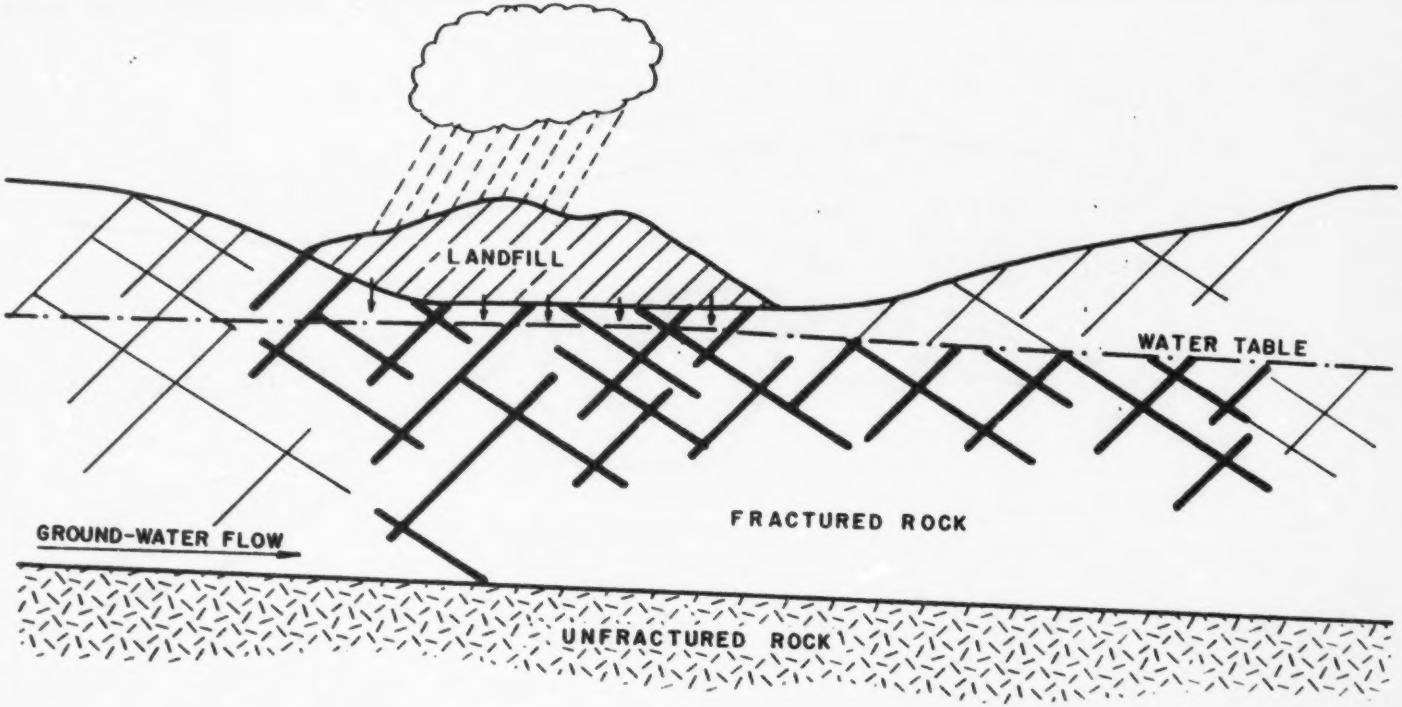

LANDFILL
WATER TABLE
FRACTURED ROCK
GROUND-WATER FLOW
UNFRACTURED ROCK

FIGURE 15. Fractured Rock Surface With A High Water Table - Leachate migrates downgradient along interconnected rock fractures to some lower natural discharge area or a pumping well.

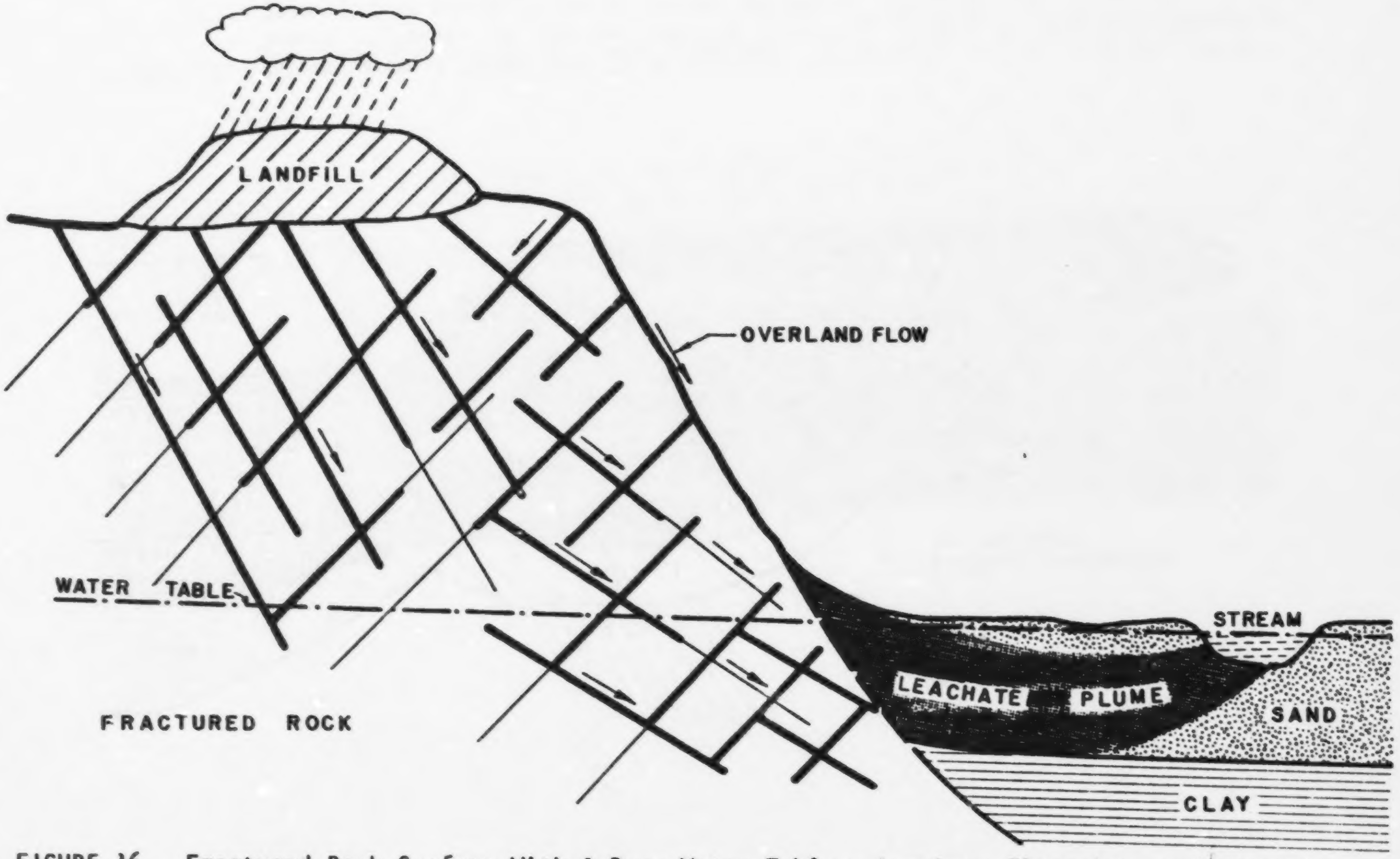

LANDFILL
OVERLAND FLOW
WATER TABLE
STREAM
LEACHATE PLUME
SAND
CLAY
FRACTURED ROCK

FIGURE 16. Fractured Rock Surface With A Deep Water Table - Leachate flows into and through interconnecting fractures and discharges either at the surface or into the subsurface where it moves with the ground water to some more distant discharge point.

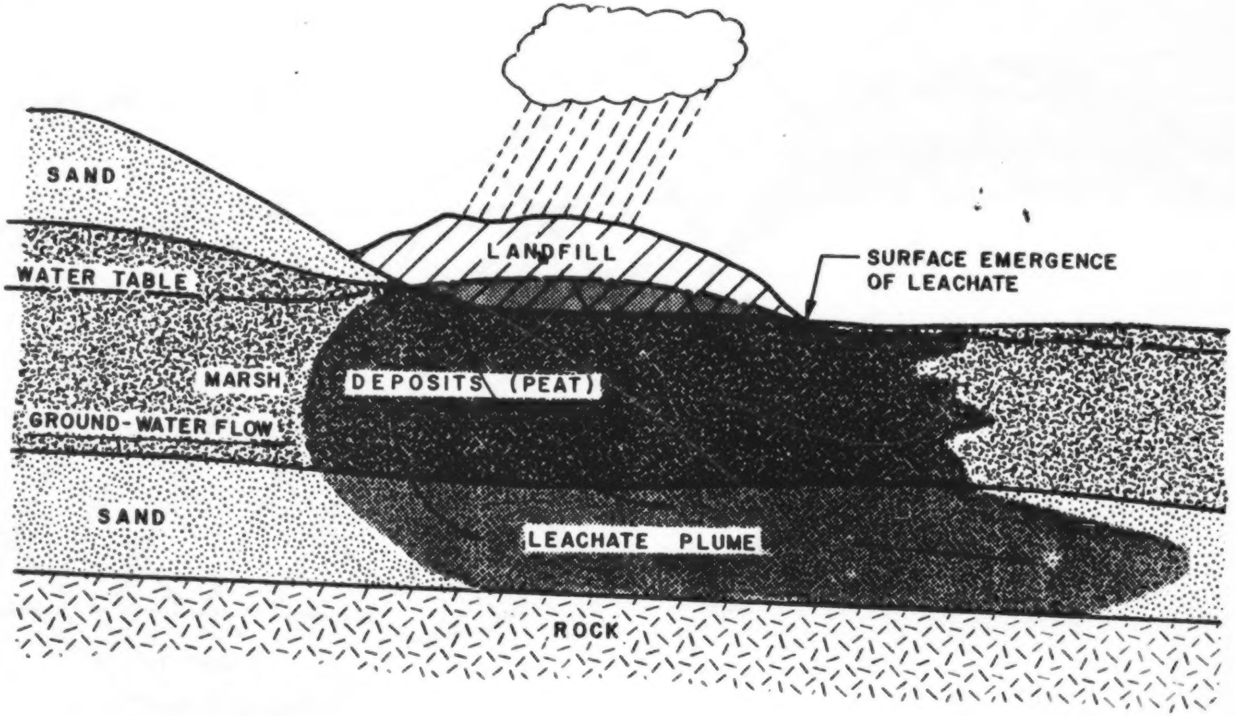

SAND
WATER TABLE
MARSH
DEPOSITS (PEAT)
GROUND-WATER FLOW
SAND
LANDFILL
SURFACE EMERGENCE OF LEACHATE
LEACHATE PLUME
ROCK

FIGURE 17. Marsh Deposit Underlain By An Aquifer - The water table is high, and a mound is formed at the base of the landfill. Leachate migrates downward through the marsh material to the aquifer. In many cases, surface emergence of leachate will occur at the toe of slope. Some contaminants may be attenuated within the marsh deposits. The portion reaching the water table moves through the aquifer with the ground water to some surface discharge point.

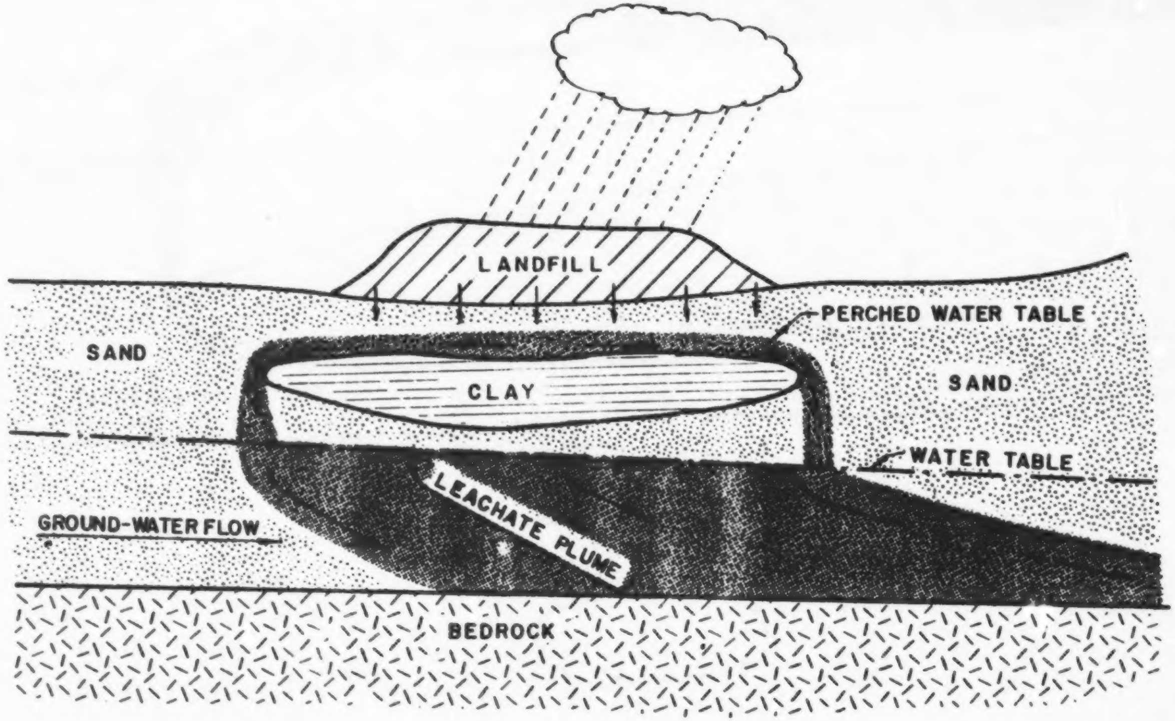

LANDFILL
PERCHED WATER TABLE
SAND
CLAY
SAND
WATER TABLE
LEACHATE PLUME
GROUND-WATER FLOW
BEDROCK

FIGURE 18. Permeable Sand Layer Underlain By A Clay Layer - The water table is deep. Leachate percolates downward under the landfill, forming a perched water table before finally reaching the actual water table.

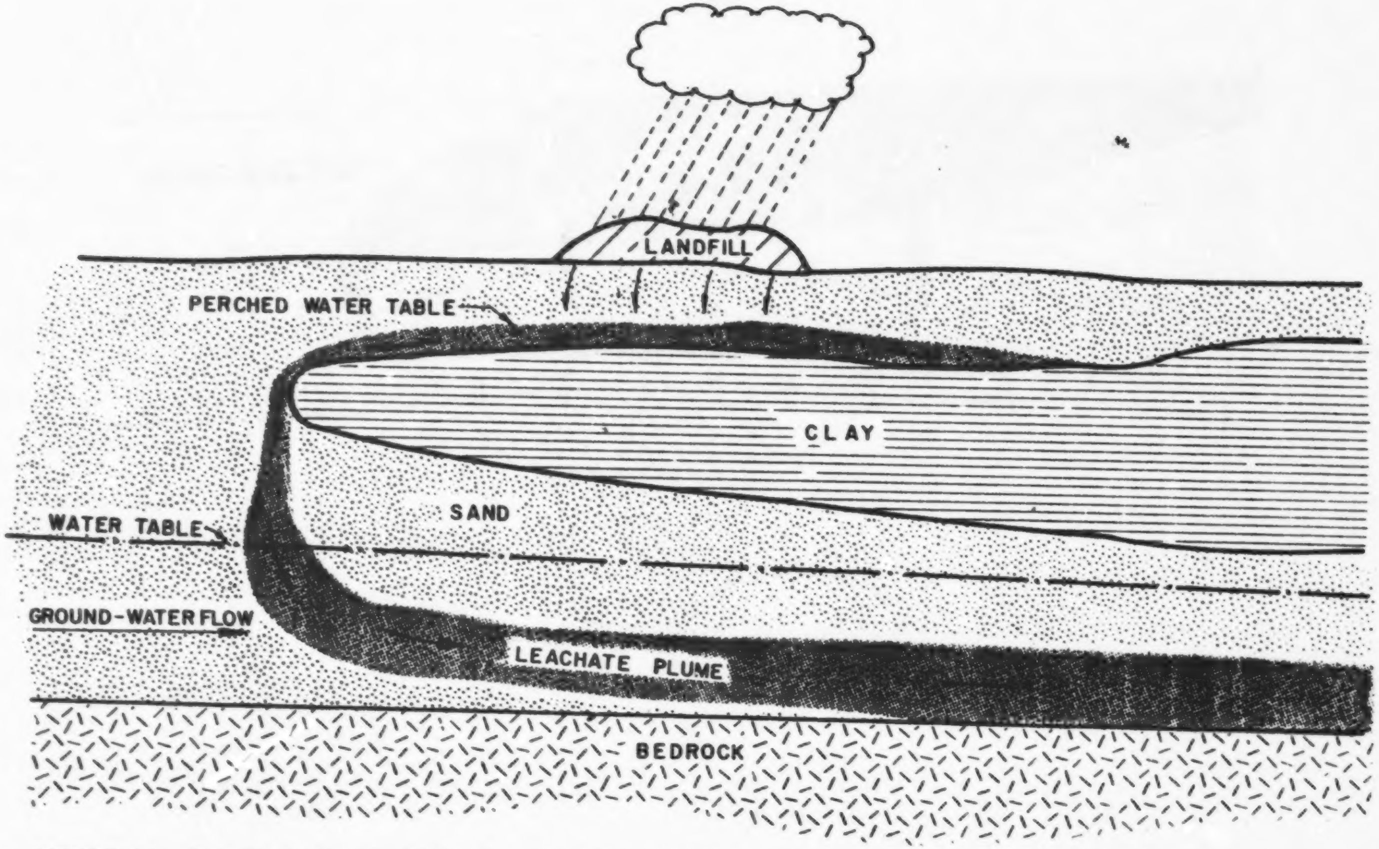

LANDFILL
PERCHED WATER TABLE
CLAY
SAND
WATER TABLE
GROUND-WATER FLOW
LEACHATE PLUME
BEDROCK

FIGURE 19 Perched Water Table Condition. Leachate percolates to the perched water water table and flows downgradient to the end of the confining layer where it moves downward to the actual water table.

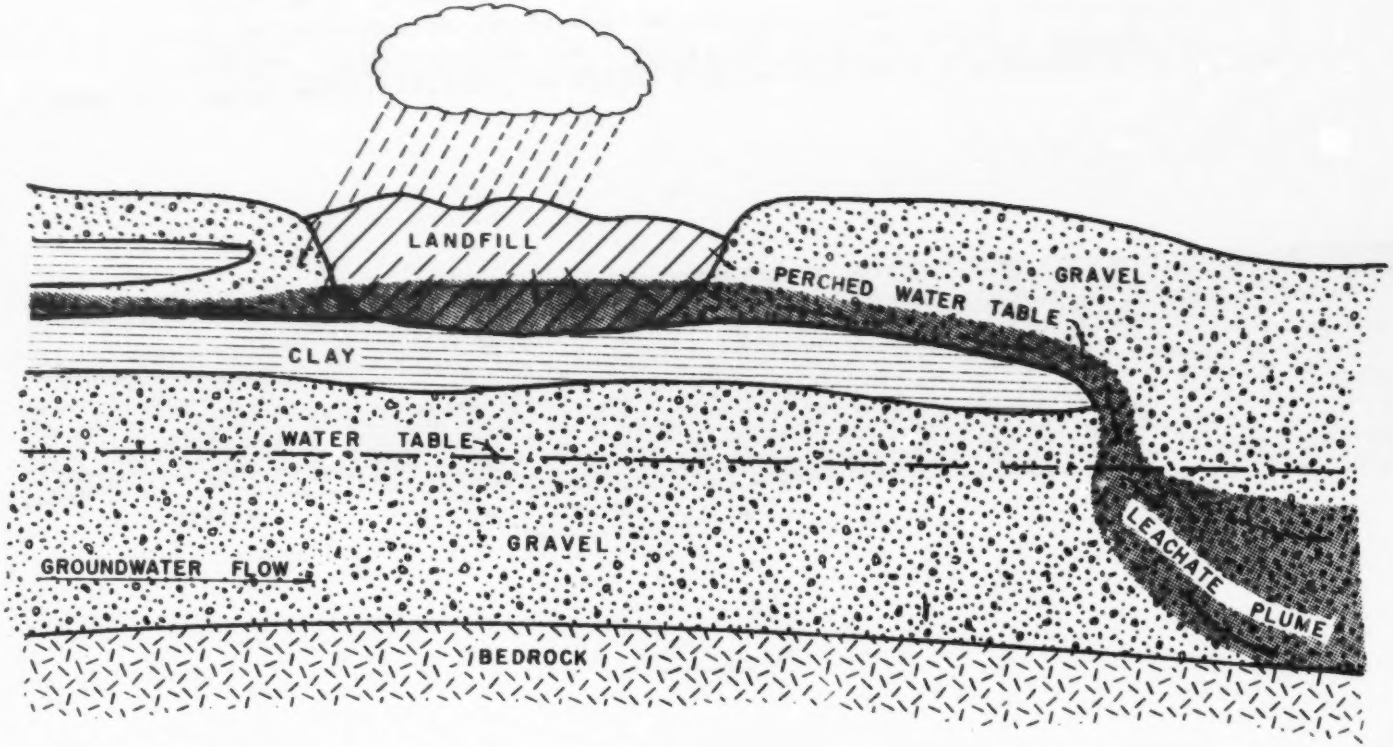

LANDFILL
PERCHED WATER TABLE
GRAVEL
CLAY
WATER TABLE
GRAVEL
GROUNDWATER FLOW
LEACHATE PLUME
BEDROCK

FIGURE 20. Abandoned Gravel Pit with a Clay Layer at its Base. A perched water table (leachate) will build up under the landfill and flow laterally through the ground above the clay until it is free to percolate to the main water table.

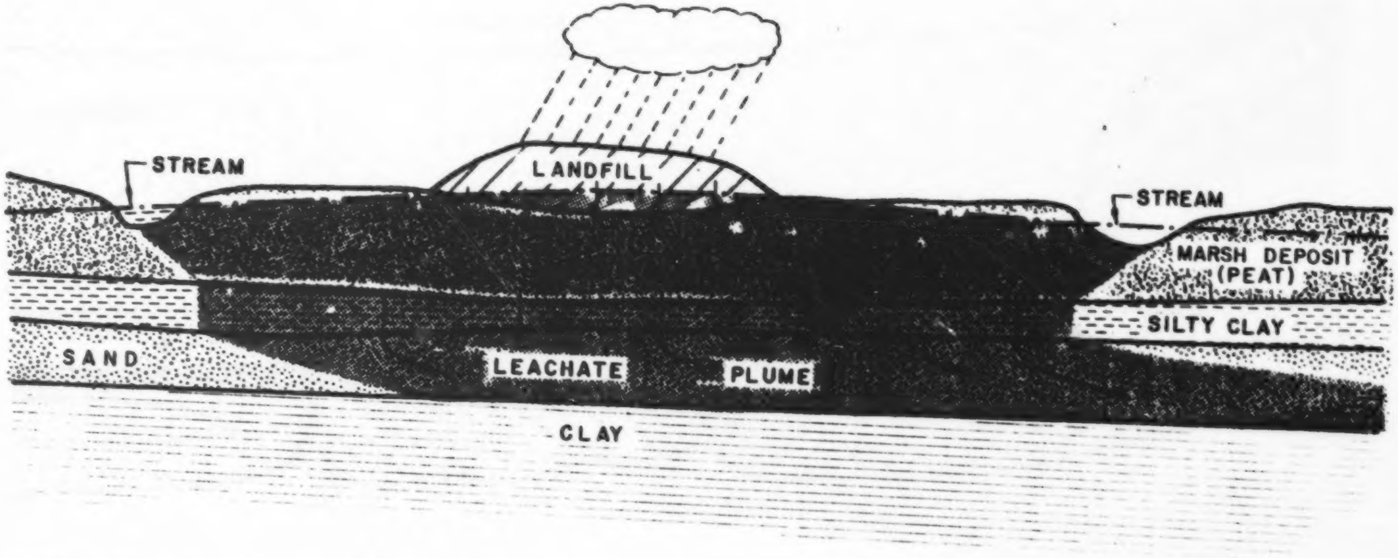

STREAM
LANDFILL
STREAM
MARSH DEPOSIT (PEAT)
SILTY CLAY
SAND
LEACHATE
PLUME
CLAY

FIGURE 21. Marsh deposits bounded on either side by streams and underlain by a shallow aquifer. Leachate from the landfill may move horizontally through the marsh materials to the stream, or vertically downward as ground-water recharge to the aquifer.

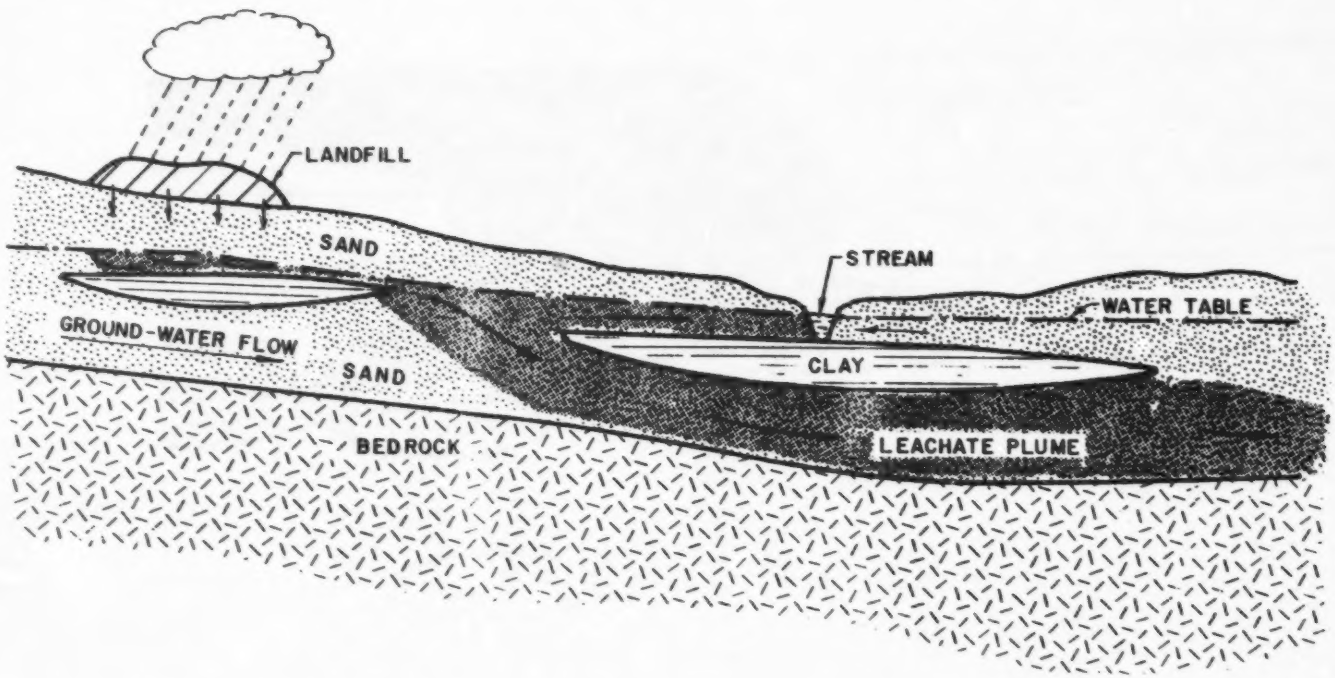

LANDFILL
SAND
STREAM
WATER TABLE
GROUND-WATER FLOW
SAND
CLAY
BEDROCK
LEACHATE PLUME

FIGURE 22. Single Aquifer Interbedded with Clay Lenses. The leachate plume is split into two plumes by a clay lens. One plume discharges into a stream near the landfill while the other plume moves deeper into the aquifer and flows to a more distant discharge point.

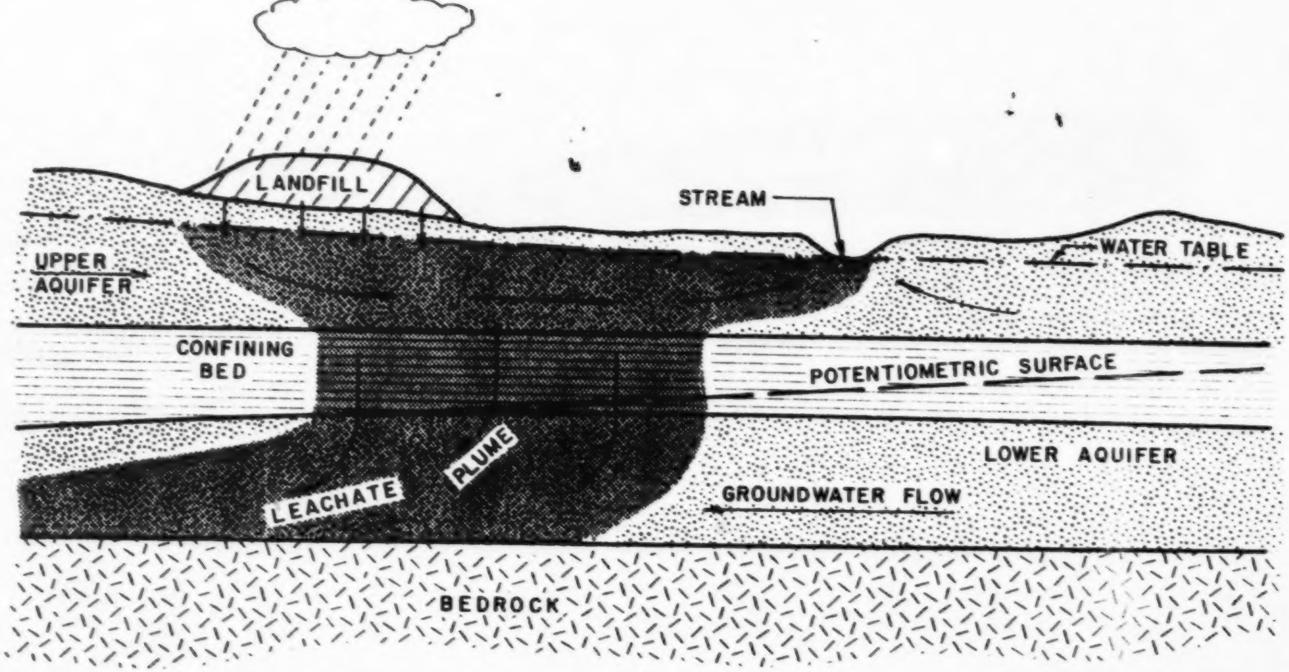

LANDFILL
STREAM
WATER TABLE
UPPER AQUIFER
CONFINING BED
POTENTIOMETRIC SURFACE
LOWER AQUIFER
GROUNDWATER FLOW
PLUME
LEACHATE
BEDROCK

FIGURE 23. Two-Aquifer System With Opposite Flow Directions. Leachate first moves into and flows with the ground water in the upper aquifer. Some of the leachate eventually moves through the confining bed into the lower aquifer where it flows back beneath the landfill and away in the other direction.

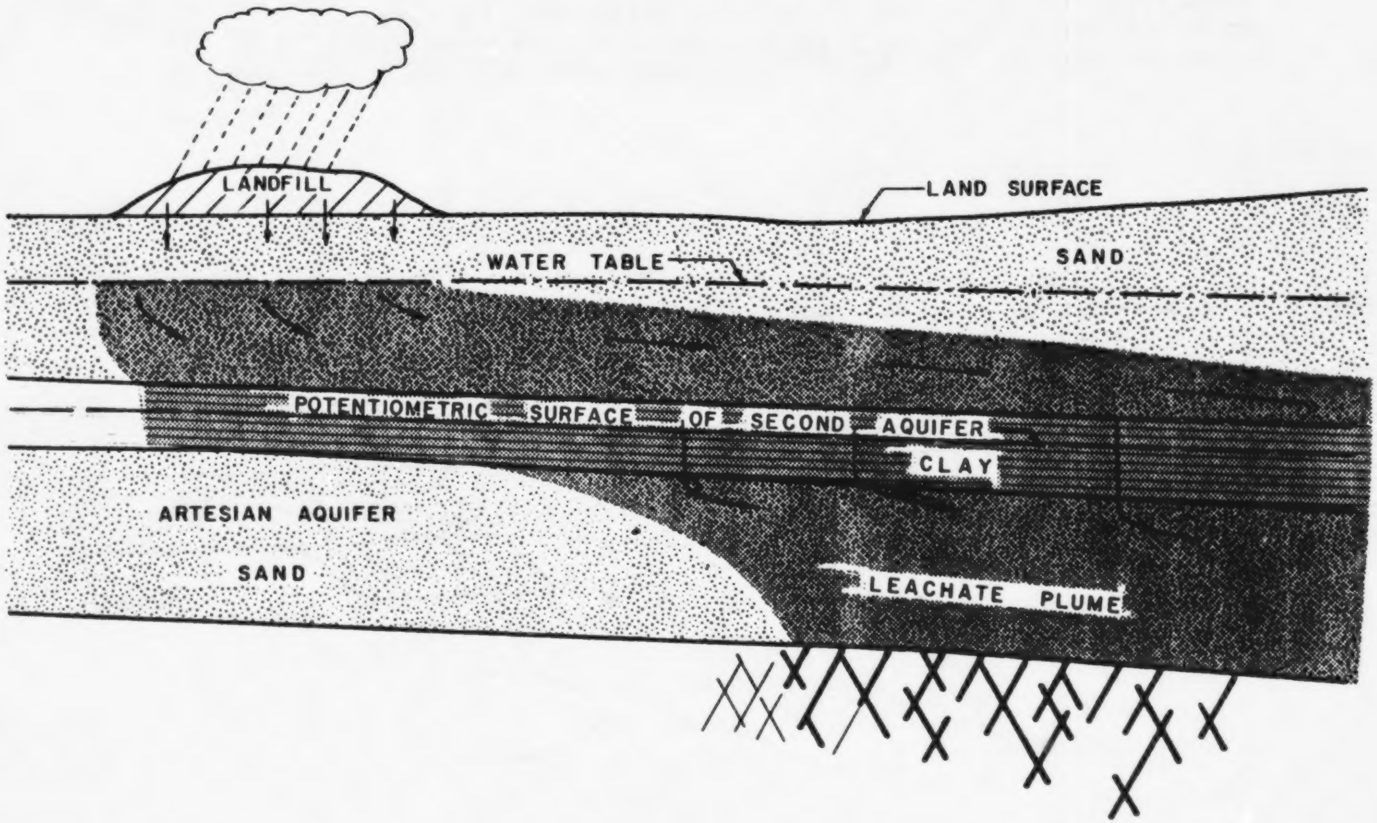

LANDFILL
LAND SURFACE
WATER TABLE
SAND
POTENTIOMETRIC SURFACE OF SECOND AQUIFER
CLAY
ARTESIAN AQUIFER
SAND
LEACHATE PLUME

FIGURE 24. Three-Aquifer System With a Deep Water Table. Leachate percolates to the upper aquifer where it moves as a plume in the direction of ground-water flow. Eventually, some of the leachate moves through the confining layer and into the second aquifer that is interconnected with a fractured bedrock aquifer.

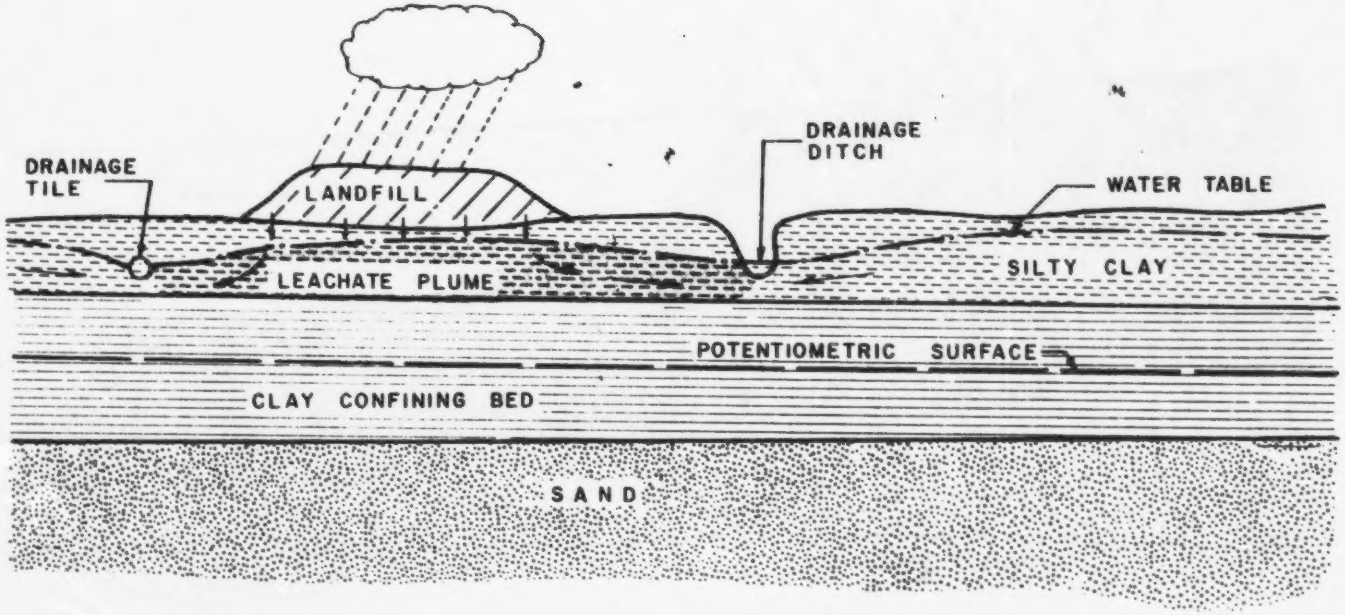

DRAINAGE TILE
LANDFILL
DRAINAGE DITCH
WATER TABLE
LEACHATE PLUME
SILTY CLAY
POTENTIOMETRIC SURFACE
CLAY CONFINING BED
SAND

FIGURE 25. Thick Clay Layer Underlain by an Aquifer. Leachate is unable to penetrate the impermeable clay layer and discharges to the tile drainage systems or drainage ditches in the area around the landfill.

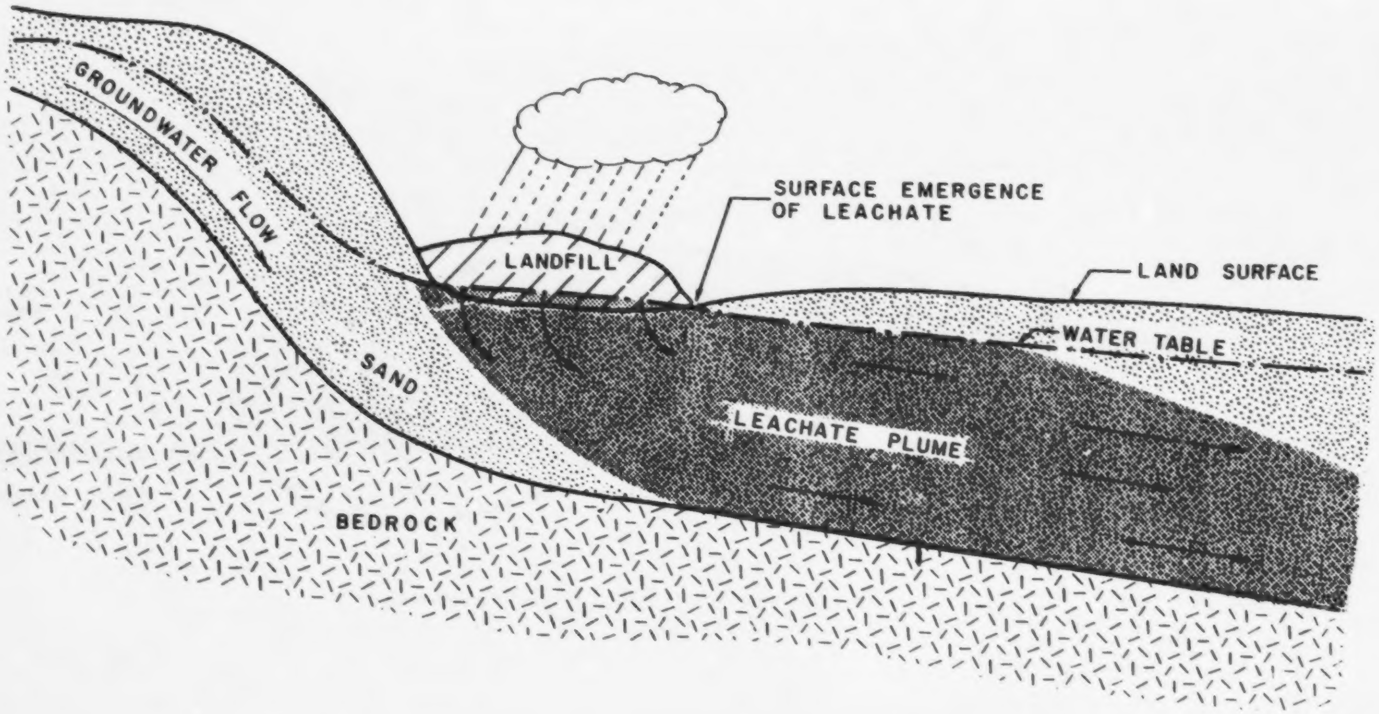

GROUNDWATER FLOW
SAND
BEDROCK
LANDFILL
SURFACE EMERGENCE OF LEACHATE
LAND SURFACE
WATER TABLE
LEACHATE PLUME

FIGURE 26. Single Aquifer Intersecting Landfill. A steep, shallow ground water table flows directly into the landfill, generating leachate which flows downward into the aquifer. Surface emergence of leachate may also occur.

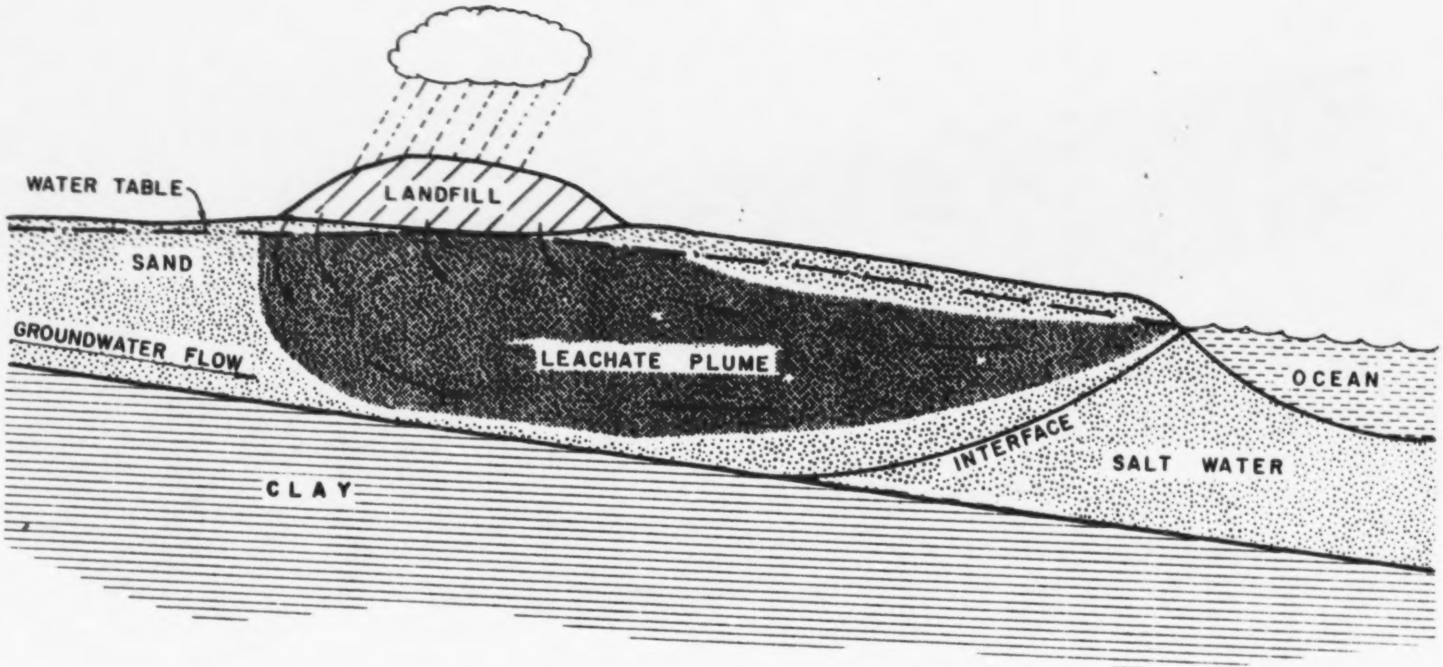

WATER TABLE
LANDFILL
SAND
GROUNDWATER FLOW
LEACHATE PLUME
OCEAN
INTERFACE
SALT WATER
CLAY

FIGURE 27. Landfill is Located Near Large Salt-Water Body. The leachate plume flows down into the fresh-water aquifer and toward the open salt-water body. As the leachate plume reaches the fresh-salt interface, it is forced upward along the interface to discharge at or near the edge of the salt-water body.

3.0 MONITORING AND WELL TECHNOLOGY

This chapter discusses and evaluates various monitoring techniques used in ground-water contamination studies relative to their use for monitoring land disposal sites. An overview of well technology is also presented to familiarize the reader with drilling methods and equipment. Cost information is presented throughout the chapter solely to demonstrate relative cost comparisons among the various monitoring techniques. Costs will vary widely with location and even with drilling contractors. Each project should establish its own costs by carefully reviewing the site conditions, the monitoring needs, and by contacting the local drilling contractors.

The monitoring techniques are divided into subsurface monitoring for the aeration and saturation zones, field inspection techniques, and other techniques such as surface-water quality measurements, aerial photography, and geophysical well logging. The monitoring purposes as well as the legal influences which are discussed in Chapter 1 will determine which one or combination of techniques to utilize. For example, if the purpose for monitoring is a state-wide evaluation of the effects of the leachate problem, a presence/probable absence determination using field inspection techniques would probably be adequate in most cases. For some larger sites, verification with some basic monitoring of the zone of saturation might also be justified. If, however, the monitoring purpose is to obtain enforcement evidence, extensive sub-surface monitoring in the zone of aeration and/or saturation would be necessary, involving the analysis of several monitoring wells.

Monitoring in the zone of aeration would not normally be part of a routine monitoring program. However, monitoring this zone is extremely useful for other than routine monitoring purposes. For example, soil analysis and pressure-vacuum lysimeter techniques are commonly used in scientific evaluations of sites; validation of design criteria, such as the soil attenuation capabilities at a site; and to check the effectiveness of an engineering design at a site. Also, legal influences may require monitoring in the zone of aeration. For example, in states where zero degradation laws are in effect, monitoring the zone of aeration has significance as an early-warning technique.

3.1 ZONE OF AERATION

The zone of aeration is that part of the earth's crust where the voids are filled with air and (usually) where some water is held by molecular attraction. It is through this zone that percolating waters must pass to recharge or contaminate the ground water. In most cases involving landfill contamination, sampling in the zone of aeration would not normally be carried out unless:

- scientific research such as measurement of attenuation is involved;
- there are unusual geologic or hydrologic considerations;
- extremely toxic chemicals are suspected in the leachate which would demand closer attention;
- sampling is to be used as an early-warning system to check the effectiveness of engineering techniques.

Such sampling is difficult, and some of the methods are expensive. However, when the decision has been made to monitor water quality in the zone of aeration, the depth to water becomes an important factor in placing the sampling devices.

Monitoring the zone of aeration is most appropriately done directly beneath the landfill where the leachate is migrating downward toward the water table. With this in mind, the monitoring devices should be placed ahead of landfill wherever possible. This avoids drilling through the landfill and the problems associated therewith.

3.1.1 Soil Analysis

Soil analysis can be valuable as a monitoring tool for tracing leachate constituents, particularly those prone to cation exchange or other adsorption reactions. Collecting soil cores beneath the landfill can be done during well installation. Techniques for core collecting are available, and methods for soil analysis are documented.[1,2,3,4,5] Soil analysis has had only limited use in leachate monitoring programs for several practical reasons. Foremost is the limited number of commercial soil-testing laboratories capable of handling soil tests outside the scope of agricultural application. For example, testing laboratories are established in each state for soil fertility analysis (nitrogen, phosphorus, potassium, pH); but heavy metals, organics, and exchangeable cations are not accommadated. In many places, only noncommercial samples are accepted by these labs.

Another restriction in soil analysis is inherent in the methodology. The total soil is seldom analyzed; only a chemical extract is considered. Fundamentally, a separation of the inorganic/organic matrix and chemical species in soil solution or "available" to soil solution must be made. An analysis of the complete soil, including the inorganic matrix,would be meaningless. The objective of the soil analyst is to measure the chemical species: in solution, exchangeable to solution, available to plants, or accumulated by adsorption or precipitation on the inorganic matrix. To meet this objective, soils must be treated with reagents of differing chemical reactivity under a variety of physical conditions. The resulting solutions are then analyzed for the chemical species of interest. Interpretation of results is a function of soil characteristics and analytical methodology. Although methods have been standardized to a degree, the analyst must be able to adapt and interpret according to the dictates of the soil sample. In contrast, water samples are usually analyzed directly or with a minimum of pretreatment. This is not to say that there are no analytical problems with water samples, but bringing a soil sample to the same state as a collected water sample involves an additional analytical step.

Soil samples yield information which cannot be obtained from water samples. Therefore, soil sampling has a place in the leachate monitoring program; its use should be expanded. Chemical species associated with soil solution, as well as those on exchange sites, can be traced downward in a soil profile or in the unsaturated zone. Locations of accumulation or leaching can be identified.

Sulfate (SO_4), chloride (Cl), and nitrate (NO_3) are soluble and unaffected by cation exchange reactions in soil. This results in mobility impeded only by the restrictions of water percolation. Soil samples can be analyzed for these anions in addition to cations which are generally more strongly associated with the solid-soil matrix. Because cations must be released from the soil matrix prior to determination, analysis is more difficult. However, zone location or heavy metals or phosphorus accumulations can only be detected through soil analysis.

In addition to the chemical information obtained from soil core sampling, valuable information on grain type and size can be gained by visual observation. Organic matter layers, clays, or silts may be encountered. Knowledge of their locations will aid in interpreting flow patterns and chemical configuration in the plume. If verification of the visual estimates is desired, a mechanical analysis can easily be made by simply separating the soil into respective proportions of sand, silt, and clay. Even more elaborate x-ray crystallographic analysis of clays will identify the minerology. This latter degree of sophistication is beyond the scope required for anything but a research program on leachate production and movement.

Decisions regarding adoption of soil sampling in a monitoring program should be made on the basis of the following criteria:

- applicability of information derived from the monitoring program;
- compliance with governmental regulations covering monitoring programs;
- relative cost of soil analysis and water analysis from the zone of aeration;
- availability of analytical facilities;
- availability of analytical techniques for the parameters of interest.

Advantages	Disadvantages
1. Ease of soil sample collection.	1. Commercial laboratories capable of non-agricultural soil analyses are scarce.
2. Accurate vertical and areal sampling locations.	2. Not a proven standardized sampling method for monitoring programs.
3. Best method to measure leachate attenuation through adsorption or precipitation mechanisms.	3. Cost of analysis likely to be higher per sample than water because of 2-step analytical procedure.
4. Long interval between sampling possible because of intermittent leachate production.	4. Applicable mainly in the zone of aeration.
5. _In situ_ conditions of sample can be maintained with proper handling.	5. Requires special equipment for each sample collection.
6. Physical and chemical condiitons throughout unsaturated zone can be observed.	6. Analytical methods not adaptable to high-rate standard procedures as available for water.
7. Samples can be stored for later comparison or further analysis.	7. Wetting and drying cycles and changes in redox potential can change chemical reactivity of some soil constituents after collection.
8. More representative biological sampling possible than with water.	8. State-of-the-art not documented in leachate studies.

3.1.2 Pressure Vacuum Lysimeters

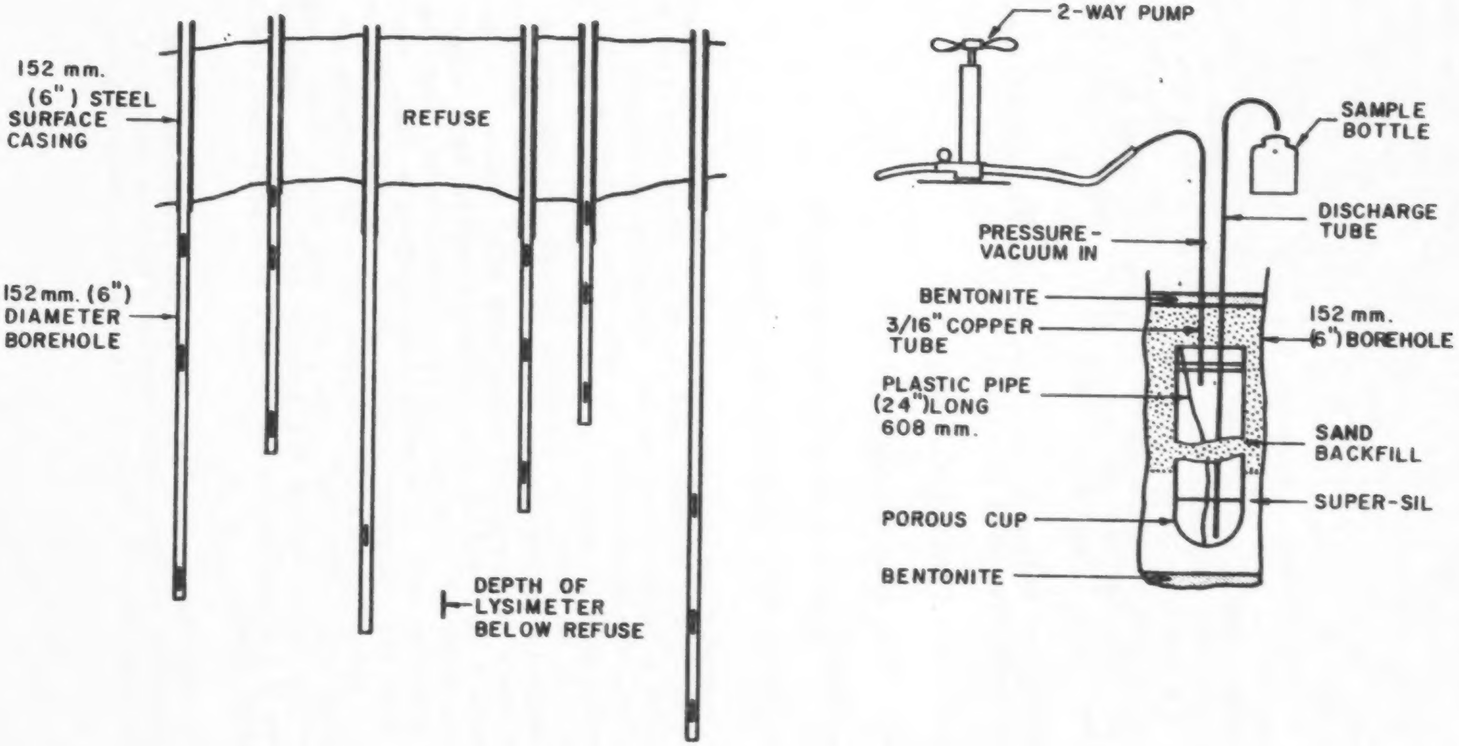
3.1.2.1 Methodology. Pressure Vacuum lysimeters have been used to obtain samples of *in-situ* soil moisture. They are used predominantly in the zone of aeration but can easily be used to sample ground water. In its most improved form, this device consists of a porous ceramic cup capable of holding a vacuum, a small-diameter sample accumulation chamber of PVC pipe, and two sampling tubes leading to the surface. Once the lysimeter is emplaced, a vacuum is applied to the cup. Soil moisture moves into the sampler under this gradient, and a water sample gradually accumulates. Then, the vacuum is released and pressure is applied, forcing the accumulated water to the surface through the sampling tube. Construction, installation, and sampling procedures are described by Grover and Lamborn (1970)[6], Parizek and Lane (1970)[7], Wagner (1962)[8], Wengel and Griffen (1971)[9], and Wood (1973)[10].

The technology of lysimeter utilization is well established. Lysimeters have been used to trace:

- pollution from septic tanks;[11]
- pollution from cesspools;[12]
- pollution from synthetic detergents;[13]
- pollution from colliery spoil heaps.[14]

Apgar and Langmuir (1971) used suction lysimeters, wells, and soil samples to study the movement and chemical characteristics of leachate from a landfill in central Pennsylvania (Figure 28).[15] At that location, the water table is more than 61 meters (200 feet) below ground surface; monitoring the unsaturated zone is of great importance. To do this, the landfill excavation was graded and lined to allow leachate to drain into a percolation trench along one side. The lysimeters were installed underneath this trench. As many as four lysimeters were emplaced at selected depths in a single borehole to a maximum depth of 16.6 meters (54.5 feet)--each installation separated from the next by a pelletized bentonite seal. Water samples collected from the lysimeter network were analyzed for Eh, pH, temperature, specific conductance, BOD, Cl^-, SO_4^-, total alkalinity, NH_3, NO_2^-, NO_3^-, PO_4^-, Ca^{++}, Mg^{++}, Na^+, K^+, and total iron. Apgar and Langmuir were able to define differences in leachate concentration from upslope and downslope cells as well as from leachate attenuation and rate of movement.

Wood (1973) suggested a modification of the lysimeters used by Apgar and Langmuir so that water samples could be recovered from any depth (Figure 29).[10] Since a check valve prevents pressurization


152 mm.
(6") STEEL
SURFACE
CASING
152 mm. (6")
DIAMETER
BOREHOLE
REFUSE
DEPTH OF
LYSIMETER
BELOW REFUSE
2-WAY PUMP
SAMPLE
BOTTLE
DISCHARGE
TUBE
PRESSURE-
VACUUM IN
BENTONITE
3/16" COPPER
TUBE
152 mm.
(6") BOREHOLE
PLASTIC PIPE
(24") LONG
608 mm.
SAND
BACKFILL
SUPER-SIL
POROUS CUP
BENTONITE

CROSS SECTION OF A LYSIMETER NETWORK
(Apgar and Langmuir, 1971)[15]

CROSS SECTION OF A TYPICAL PRESSURE-VACUUM LYSIMETER INSTALLATION
(Parizek and Lane, 1970)[7]

FIGURE 28. PRESSURE-VACUUM LYSIMETER EMPLACEMENT BENEATH A LAND DISPOSAL SITE

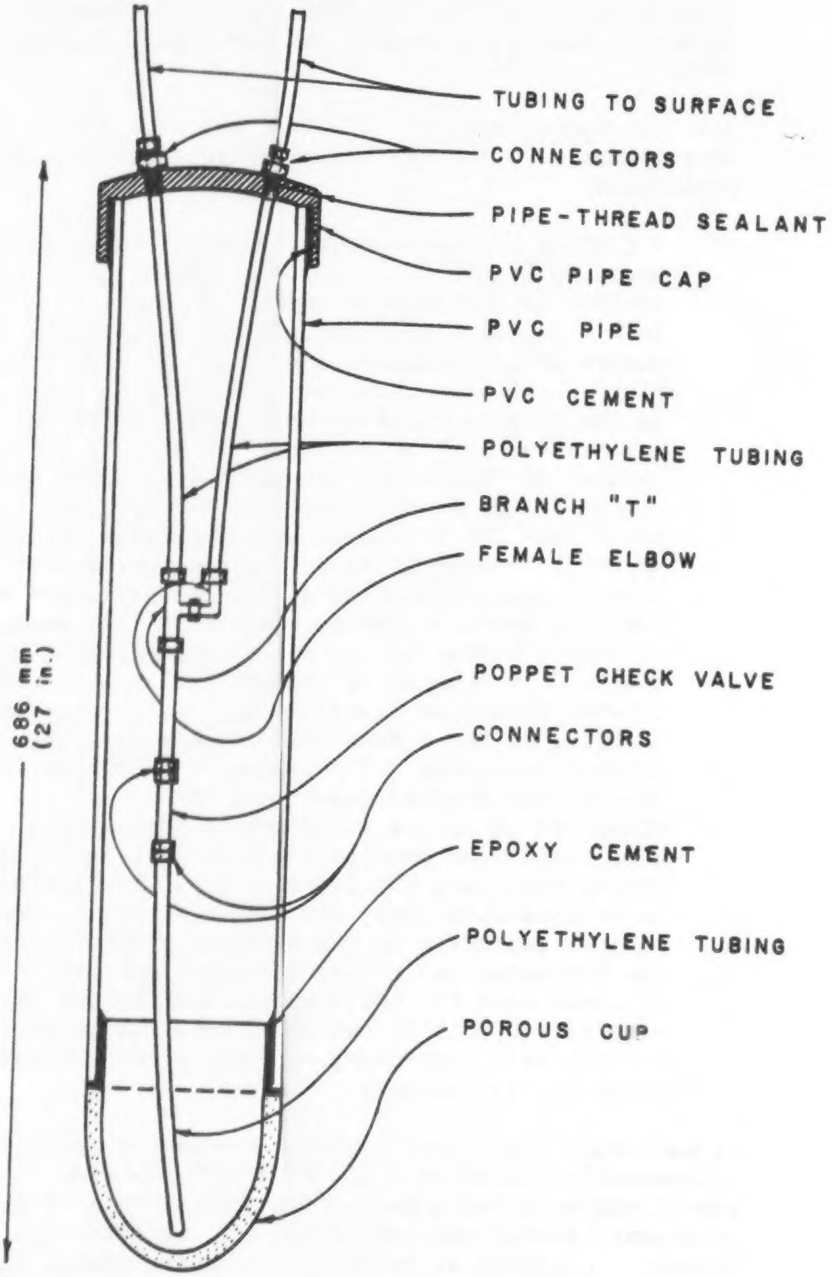

TUBING TO SURFACE
CONNECTORS
PIPE-THREAD SEALANT
PVC PIPE CAP
PVC PIPE
PVC CEMENT
POLYETHYLENE TUBING
BRANCH "T"
FEMALE ELBOW
POPPET CHECK VALVE
CONNECTORS
EPOXY CEMENT
POLYETHYLENE TUBING
POROUS CUP
686 mm
(27 in.)

FIGURE 29. MODIFIED PRESSURE-VACUUM LYSIMETER INSTALLATION
(After Wood, 1973)[10]

of the porous cups, deep pressure-vacuum lysimeters appear to be the best method of monitoring the zone of aeration. In deeply placed lysimeters, pressure exceeding about one atmosphere in the sample chamber would drive accumulated water back through the cup rather than to the surface.

3.1.2.2 Implementation. Parizek and Lane (1970) have given a detailed description of pressure-vacuum lysimeter installation and sampling procedures:

> A typical pressure-vacuum lysimeter installation is shown in Figure 29 (sic.). Placement holes are first drilled to the desired depth. They may be 101 to 152mm (4 - 6 inches) (sic) in diameter depending upon the number of lysimeters to be placed in each hole. A plug of wet bentonite clay is placed in the bottom of the hole to isolate the lysimeter from the undisturbed soil below it. This plug is optional. A layer of "Super Sil" at least six inches deep, is placed on top of the bentonite. "Super Sil" is the trade name for a commercially available, crushed, pure silica-sand of almost talcum powder consistency. This is used to provide a clean transmission medium for soil moisture moving under capillary pressure, to insure hydraulic contact of the adjacent soil medium with the porous tip, to fill uneven voids created during drilling, as well as to discourage clogging of the ceramic tip by colloids, organic matter, or soil particles. The lysimeter is placed in the hole to the desired depth, and "Super Sil" is placed around it until the lysimeter is half buried. Native soil, free from pebbles and rocks, is backfilled and tamped with long metal rods. After the lysimeter is covered with about six inches of soil, a second plug of bentonite is deposited to further isolate the lysimeter and to guard against possible channeling of water down the drill hole. Backfilling is continued with native soil to the depth where it is desired to set the next lysimeter, at which point the above procedure is repeated.

It was found that three lysimeters were the maximum that could be conveniently placed in a 152-mm(6-inch)diameter hole. If more than three were installed, difficulties arose in proper depth placement, proper tamping of backfill material was prevented, the danger of crimping or tangling the copper tubing was increased, and the risk of channeling soil water down the incompletely filled hole became greater. Care was taken to accurately measure the depth of placement of each lysimeter. It was possible to set the lysimeters to within 152mm (6 inches) of the desired depth even in 9-meters (30-foot) deep holes.

After the lysimeters are placed, a short section of flexible Tygon plastic tubing is secured over the end of each copper access tube with PVC electrical tape. This procedure allows thumb-screw pinch clamps to be used to seal the lysimeter between sampling periods, thereby maintaining the vacuum within the lysimeter.

The pump used in conjunction with these pressure-vacuum lysimeters is a two-way hand pump that can either deliver a back pressure or pull a vacuum. It is similar to a tire pump and can be purchased from any laboratory equipment supply house. A small vacuum gauge may be installed on the pump by means of a tee union. This enables the operator to consistently apply a desired vacuum (about 457mm--18 inches--of mercury) to all lysimeters. A length of Tygon tubing is secured to each of the pump's pressure and vacuum parts, allowing the pump to be coupled to the access tubes of the lysimeters. The free ends of the pump's tubing are slipped over a short length of copper tubing that is secured to the pressure-vacuum tube of the lysimeter and is held securely by a small spring-loaded clamp.

A typical pressure-vacuum lysimeter sampling sequence is as follows:

- The lysimeter's discharge tube is clamped shut and the vacuum side of the two-way pump is attached to the "in" tube.
- A vacuum of approximately 457mm (18 inches) of mercury is drawn, and the "in" tubing is clamped shut.
- To recover soil-water samples, the pressure side of the two-way pump is attached to the lysimeter's "in" tube; and the pinch clamps are removed. A few strokes of the hand pump generates enough pressure to force the water out of the lysimeter and into a collection bottle placed under the discharge tube.
- After emptying the lysimeter, the vacuum side of the pump is attached to the "in" tube, and the lysimeter is evacuated again to gather another sample.

Advantages	Disadvantages
1. Inexpensive sampling device of great reliability.	1. Considerable care must be taken during operation. A person with adequate training in the procedure is required.
2. Inexpensive installation.	2. Sampling device failure is irreparable.
3. Standard water analysis can be made.	

4. Samples can be collected at a central point.	3. Small volume of sample, limiting number of analyses that can be made.
	4. Surface tubing subject to tampering unless adequately protected.
	5. Use at depth greater than 33 meters (108 feet) not documented.
	6. Potential sample contamination by porous cup if material is not properly prepared.
	7. Possible plugging of cup by colloidal materials; cup might exclude large molecules and alter the quality of the sample.

3.1.3 Trench Lysimeters

3.1.3.1 Methodology. Several investigators have used trench lysimeters to sample water from irrigation or rainfall near the surface of the zone of aeration. In normal practice, a wood-reinforced trench or concrete-ring caisson is installed to a depth of 3 to 9 meters (10-30 feet) below land surface. Pans,[7] troughs,[16] or open end pipes,[12] are forced out of the trench (caisson) through access ports into the subsoil. These collecting devices intercept percolating water and conduct it to sample bottles inside the trench. Only after irrigation or precipitation is there sufficient water infiltrating the subsoil to collect a sample.

Due to the potential accumulation of hazardous gases generated by decomposition of landfilled material, the use of an open trench or caisson to collect leachate for sampling can be dangerous. Artificial ventilation and gas monitoring devices are required to prevent injury to personnel recovering samples collected inside the trench.

3.1.3.2 Implementation. A description of a typical trench lysimeter is excerpted from Parizek and Lane (1970).[7]

> A 1.2 meter (4-foot) wide, 3.6 meter (12-foot) long (sic) trench was excavated to a depth of 3 meters (10 feet) (sic) using a back hoe. The hoe was braced with timbers and siding to allow safe access to the trench. The trench was then hand-dug to a 5.2 meter (17-foot) (sic) depth and braced. The entire seepage face was inclined

> 1 to 5 degrees from the vertical and sloped toward a hill down which soil water and interflow was expected. The residual soil contained resistant chert and quartzite cobbles and boulders and was reasonably well cemented with iron oxide and clay. As a result, the pans could not be inserted into the soil profile without first providing an opening. A sheet metal blade 102-mm (4-inches) (sic) wide and .6-meters (2-feet) (sic) long was hammered into the overhanging bank with a sledge hammer to provide access for the pans. Pan lysimeters were tapped into these openings and allowed to slope gently toward the trench. Voids above and below the pans were backfilled with soil, tamped into place. As siding was added to the trench walls, holes were cut to allow the copper tubings to project into the sampling pit. Spaces between the original trench faces and siding were filled with native soil and washed pea-gravel to allow water to flow freely toward the pit floor. After the walls and braces were emplaced, plastic tubing was connected to the copper tubing and inserted into plastic sampling bottles. The sampling pit was covered with a sloping roof and a half-round drain pipe was used to divert roof water away from the installation. A ladder was placed at one end of the house to allow access.

Alternate methods of construction are to place concrete manhole rings in an open excavation or to sink them to a desired depth using caisson construction techniques. The decision as to which method to follow would depend upon soil stability.

Advantages	Disadvantages
1. No apparent advantages.	1. Collection procedure dangerous because of possible flamable gas accumulation in trench.
	2. Samples can be collected only after rainfall.
	3. Construction of trench or caisson is expensive.
	4. No documentation of application to landfills.

3.2 ZONE OF SATURATION

In the zone of saturation, leachate movement from a landfill will be controlled by a combination of ground-water flow patterns and soil-leachate interactions. Under shallow water-table conditions, only a thin zone exists where unsaturated soil-leachate interactions can reduce leachate concentrations. Therefore, careful collection of representative ground-water samples from properly constructed wells is necessary to trace leachate movement or to determine its presence in the ground-water environment.

3.2.1 Well Screened or Open Over a Single Vertical Interval

3.2.1.1 Methodology. Wells screened over a single vertical section of an aquifer are the most common construction method used to obtain ground-water samples from unconsolidated sediments or semi-consolidated rocks. Uncased wells (open hole) in consolidated rock can be used for the same purpose. Although this type of well is routinely used in monitoring leachate contamination of the ground water, a single well is not effective in providing information on the vertical distribution of a contaminant.[17,18]

In practice, a well is drilled to an arbitrary depth, usually just below the water table in landfill studies. The screen is positioned to intersect the water table as shown in Figure 30. The rationale for this type of construction is that if leachate reaches the ground water, it will be detected in water samples from this type of well. The drawback of the construction is immediately apparent; only a portion of the aquifer is sampled, and only the most recently infiltrated leachate can be collected. In most cases, leachate will be under partial control of a density gradient and will sink into the ground-water body. This denser fluid body, sinking into fresher water, cannot be sampled with a well that skims only the top of the water body. However, great reliance has been placed on this type of well construction to trace the extent of leachate movement into an aquifer.

Although a well may be completed below the water table, it may not provide water samples representative of leachate concentration at that point. For example, the well casing may entirely seal off the contaminated aquifer; or the screen may penetrate into another aquifer system (if little is known about site geology). The information gained from these water samples would be misleading. This drawback can be counteracted in part when the well is screened over the entire aquifer thickness. However, if the aquifer is thick and the contaminated plume is thin, the composite ground-water sample obtained provides no information on the vertical distribution of leachate; sample dilution may mask the contamination.

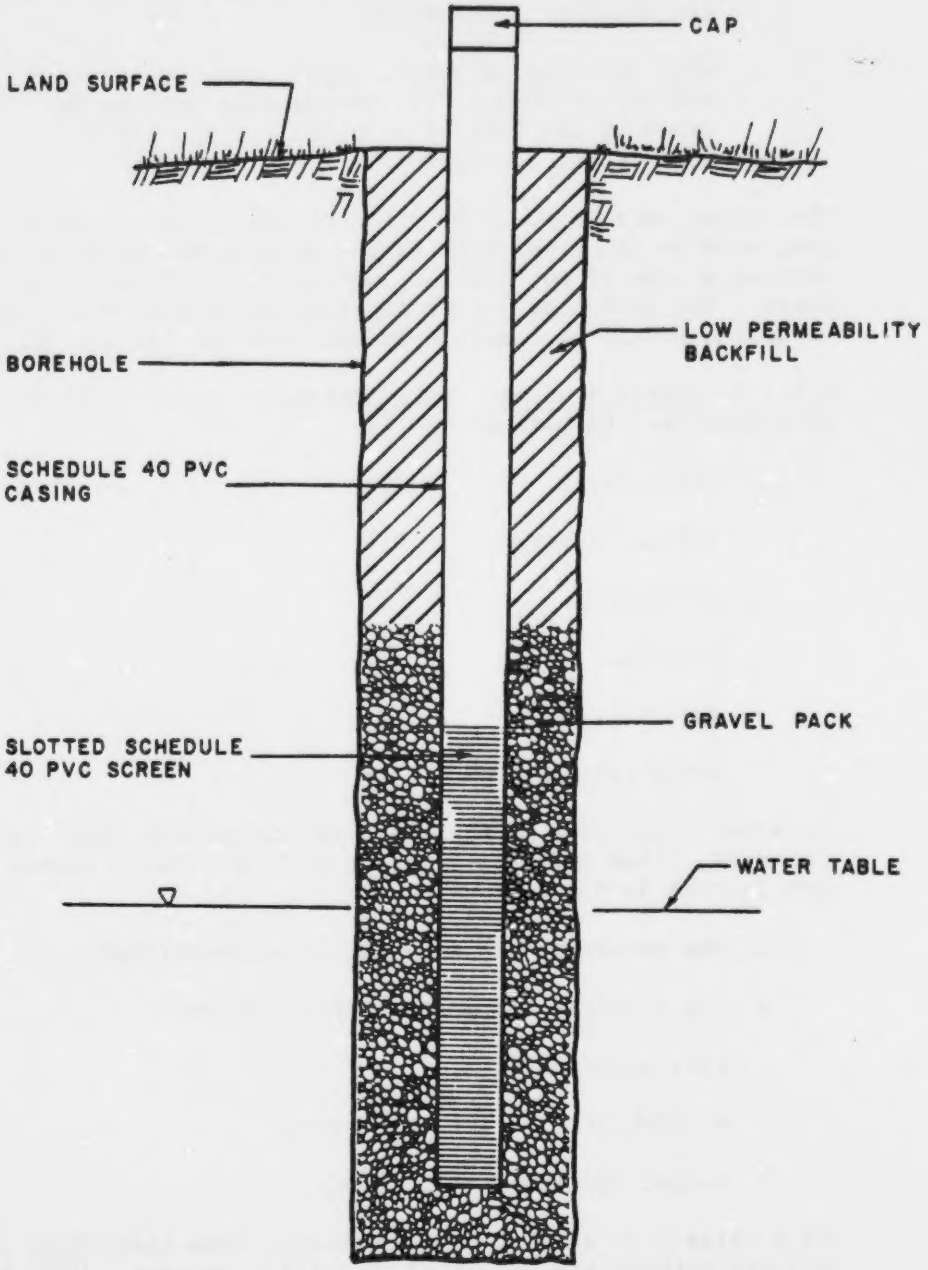

CAP
LAND SURFACE
LOW PERMEABILITY BACKFILL
BOREHOLE
SCHEDULE 40 PVC CASING
GRAVEL PACK
SLOTTED SCHEDULE 40 PVC SCREEN
WATER TABLE

FIGURE 30. TYPICAL MONITORING WELL SCREENED OVER A SINGLE VERTICAL INTERVAL

Taking everything into consideration, using a single-screen well appears to be justified under two situations:

- when obtaining composite ground-water samples from wells in which the entire saturated thickness of the aquifer is screened;
- when the sampled area's depth to water is great, and the major part of the sampling program is aimed at the zone of aeration and the top of the zone of saturation.

The latter case is probably the best use of this type well. The wells, completed in the upper zone of the water body, would serve as an early-warning system if any leachate were able to percolate to the ground water. Once detected, other sampling techniques would be required to trace the extent of leachate and movement in the aquifer.

3.2.1.2 Implementation. This type well can be drilled by a variety of techniques, including:

- mud-rotary,
- reverse-rotary,
- air-rotary,
- jetting,
- augering,
- drive points.

Diameters range from 32mm (1¼ inches) to greater than, but rarely exceeding, 914mm (36 inches). The drilling method chosen depends upon factors such as:

- the nature of the material to be penetrated;
- the diameter and depth of well desired;
- site accessibility;
- availability of drilling water;
- budget and time constraints;

and a variety of other factors resulting from individual site conditons. Drilling methods are discussed in a later section. With the possible exception of hand augering and the installation of drive points, a drilling contractor should be hired to install this type of well, unless the investigator has access to a power auger, soil boring, or jetting rig.

A hypothetical water-table aquifer has been assumed for cost comparisons of the various techniques described in this chapter. This aquifer consists of sand, with a depth to water of 3 meters (10 feet) and a saturated thickness of 30 meters (100 feet).

As mentioned in Chapter 2, the recommended type of construction for monitoring purposes is to screen the well over the entire saturated thickness of the aquifer. (The example is 30 meters—100 feet.) The quickest and least expensive way to complete this type of installation would be to:

- drill a 152- to 203-mm (6 to 8-inch) diameter borehole with a hydraulic rotary rig to the bottom of the aquifer;

- set 102-mm (4-inch) diameter slotted PVC well screen and PVC casing;

- backfill with a gravel pack or formation stabilizer;

- place a concrete collar around the well casing at ground surface to prevent downward leakage of rainwater or other fluids.

For the hypothetical example, the cost of this installation would be in the range of $2,300 to $4,500 for drilling, materials, installation, and development. A well screen is second only to drilling in terms of cost, ranging from $1,000 to $1,500 for 30 meters (100 feet) of 102-mm (4-inch) slotted PVC. Construction cost could be reduced to a total of $1,600 to $3,700 if 51-mm (2-inch) casing and screen are used; but sampling can be more difficult and tedious in a well of this diameter. On the other hand, using 152-mm (6-inch) casing and screen facilitates development and water sampling but increases the cost to the range of $6,400 to $7,500 per well. In wells of this size, wire-wound or louvered metal well screens are more commonly used than PVC, resulting in a substantial cost increase per installation as compared to the 102-mm (4-inch) well.

If the investigator is interested in sampling only the top of the aquifer with a well constructed so that a 3-meter (10-foot) long, 102-mm (4-inch) diameter screen intersects the water table, price per installation would range from $700 to $1,200. This is a substantial reduction in the expenditure required to monitor the ground water; but as discussed above, it is not a completely reliable technique for assessing ground-water contamination by leachate. Even greater reductions in expenditure per installation can be obtained by installing a 51-mm (2-inch) diameter, 1.5-meter (5-foot) long drive point by hand--the total cost of which would be less than $200 per well including labor, materials, and development with a pitcher pump.

Advantages	Disadvantages
1. Small diameter, shallow wells are quick and easy to install.	1. No information is given on the vertical distribution of the contaminant.
2. Can provide composite ground-water samples if screen covers saturated thickness of aquifer.	2. Improper completion depth can cause error in determining leachate distribution.
3. Can be drilled by a variety of methods.	3. Screening over much of the aquifer thickness can contribute to vertical movement of contaminant.
	4. Leachate may become diluted in the composite sample, resulting in lower than actual concentrations.

3.2.2 Piezometers

3.2.2.1 Methodology. The terms piezometer and observation well are commonly used interchangeably; however, there is a significant difference between them. As implied by its name, piezometer is a pressure measuring device, frequently used for monitoring:

- water pressure in earthen dams or under foundations;
- water pressure in aquifers.

In the first instance, the piezometer resembles a porous tube or plate; in the later, it resembles a screened well or open hole. A piezometer is isolated from other pressure environments by an impermeable clay or cement seal. Water samples representative of a specific horizon can be obtained from well-type piezometers--a highly desirable factor in designing a monitoring program (Figure 31).[19] If the well screen is properly isolated by an impermeable seal placed immediately above the screen, a piezometer can also be used to measure vertical head differences under unconfined conditions. Any well constructed without this seal cannot be considered a piezometer. In application to landfill leachate monitoring, there is a significant difference

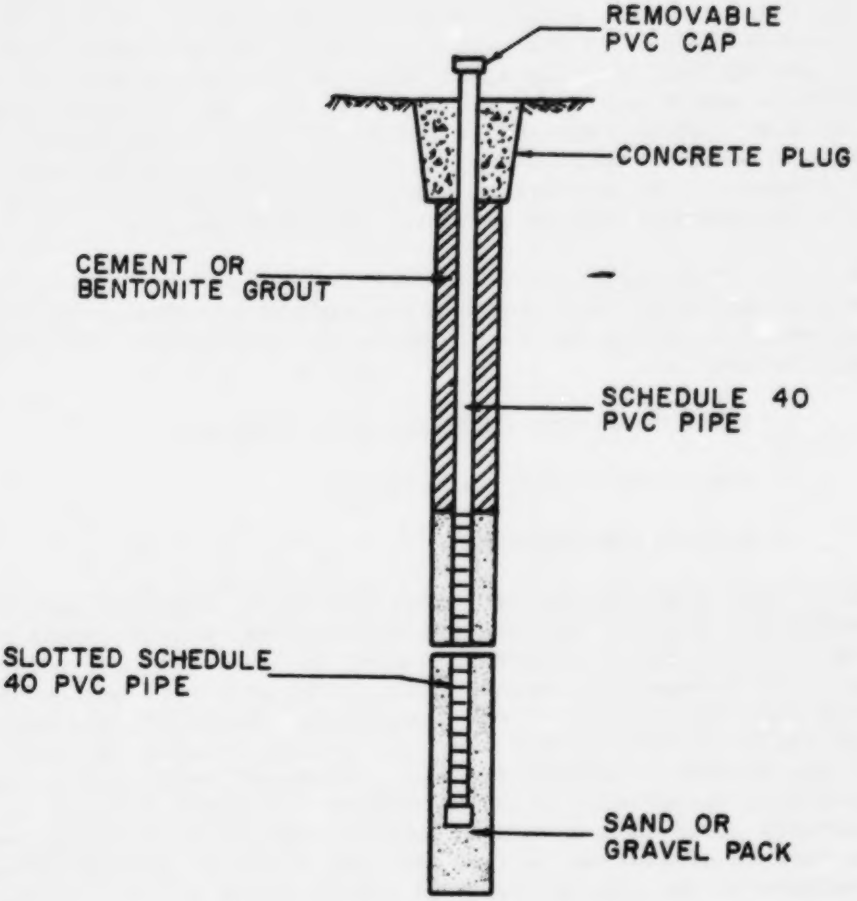

REMOVABLE PVC CAP
CONCRETE PLUG
CEMENT OR BENTONITE GROUT
SCHEDULE 40 PVC PIPE
SLOTTED SCHEDULE 40 PVC PIPE
SAND OR GRAVEL PACK

FIGURE 31. PIEZOMETER WELL INSTALLATION FOR SHALLOW GROUND-WATER MONITORING

(After Clark, 1975)[19]

between a piezometer and a well screened over the entire vertical interval. The relatively impermeable annular seal will prevent downward movement of leachate into uncontaminated zones of the aquifer.

A low-cost modification of a typical engineering piezometer will allow collection of in-situ ground-water samples throughout the saturated thickness of an aquifer. The piezometer, on modification, resembles the deep pressure-vacuum lysimeter (Figure 32). Porous PVC is used instead of a ceramic cup which in this instance is unnecessary and would decrease the effectiveness of the sampler. With porous PVC, a vacuum applied to the sampling chamber is immediately transmitted to the aquifer, drawing water into the chamber; a porous ceramic cup holds a vacuum and water slowly moves into it. Using porous PVC (or even slotted PVC) should reduce the cost of the sampler below the approximate $30 cost of commercially available deep pressure-vacuum lysimeters. The surface sampling procedure can be the same as that for the pressure-vacuum lysimeter described earlier.

3.2.2.2 Implementation. To place a grout or bentonite seal around a well casing as required in piezometer construction, an annulus between the casing and the borehole wall is needed. Drilling methods are limited to:

- cable tool with inner and outer casing;
- one of the rotary techniques;
- hollow stem augering.

After casing and screen have been installed, a gravel pack is placed around the screen. To seal the well casing, a neat cement grout or bentonite slurry is poured or pumped into the annulus, preventing the vertical leakage that might occur if the well were merely backfilled with cuttings or fill. From a sampling standpoint, the seal is vital. The sample withdrawn from the well is from a known vertical interval of the aquifer. Without the seal, rainwater would infiltrate the backfill, potentially diluting samples collected from the well; or leachate could move downward, causing samples to be unrepresentative. Another consideration is that the seal tends to prevent movement of leachate in the annular material, which could act as a conduit to uncontaminated zones of the aquifer. Constructing a monitoring well that contributes to or hastens the spread of contamination is obviously not a recommended procedure.

Since piezometers and wells screened over a single interval are identical (except for an impermeable seal between the casing and borehole wall), the only price difference is that incurred for placing the impermeable seal and purchase of necessary sealing

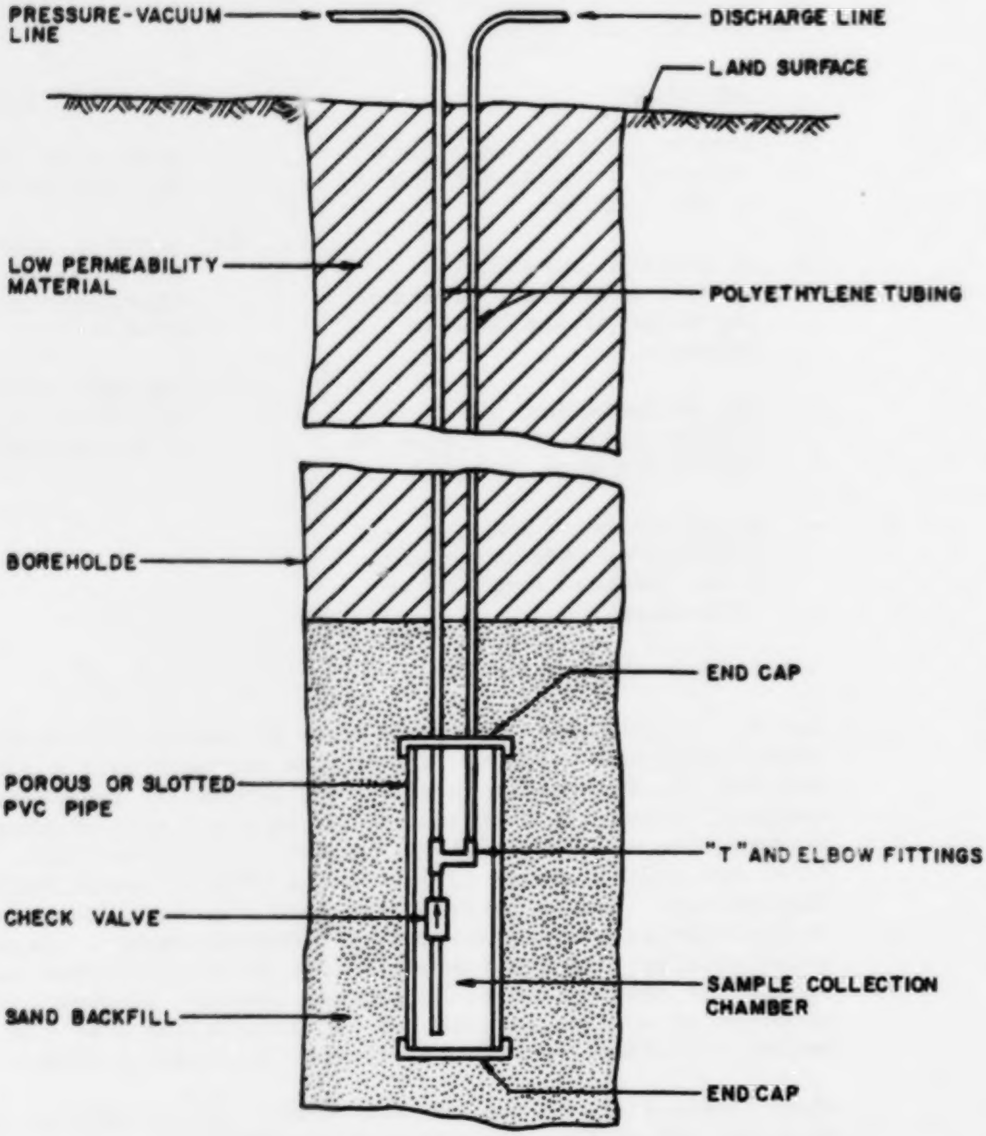

PRESSURE-VACUUM LINE
DISCHARGE LINE
LAND SURFACE
LOW PERMEABILITY MATERIAL
POLYETHYLENE TUBING
BOREHOLDE
END CAP
POROUS OR SLOTTED PVC PIPE
"T" AND ELBOW FITTINGS
CHECK VALVE
SAMPLE COLLECTION CHAMBER
SAND BACKFILL
END CAP

Figure 32. Details of a Low-Cost Piezometer Modified For Collection of Water Samples

materials. In the hypothetical aquifer, only a 3-meter (10-foot) seal is required. A two-man crew should be able to emplace it in half a day to a day. This would increase installation cost about $500 to $1,000 if grout were pumped into place and approximately $250 to $450 if bentonite were placed by hand. Total estimated cost per installation can be found on Table 3.

Advantages	Disadvantages
1. Sample is collected from a selected vertical section of the aquifer.	1. Restricted number of drilling methods.
2. If properly constructed, technique prevents downward migration of leachate in borehole.	2. Improper completion depths can cause error in determination of leachate distribution.
3. Can be installed inexpensively and rapidly if casing diameter is small.	3. Improper construction can contribute vertical migration of contamination.
4. Modification of an engineering piezometer will allow vertical sampling of contaminant.	

3.2.3 Well Clusters

3.2.3.1 Methodology. As previously discussed, the major drawback in using individual wells or piezometers screened over a single vertical interval of the aquifer is that they provide no information on the vertical distribution of the contaminant and only rudimentary information on its area distribution. To overcome this, investigators have used well clusters to define the vertical distribution of a contaminant. Each cluster consists of a group of closely-spaced, small-diameter wells completed at different depths. From these wells, water samples that are representative of the different horizons within the aquifer can be collected. Careful placement of well clusters at a landfill site and the surrounding area will allow reliable delineation of both vertical and areal leachate distribution.

Well clusters are by far the most common and successful technique to date for delineating ground-water contamination. One shortcoming, however, is that regardless of the selection of the completion depth of each well in the cluster, there are unsampled intervals through which leachate may pass undetected. Several approaches to selecting depth have been made:

TABLE 3

COST ESTIMATES FOR VARIOUS MONITORING TECHNIQUES AND CONSTRUCTION METHODS IN THE ZONE OF SATURATION

Monitoring Technique & Construction Method	Price Per Installation Well Diameter		
	51mm (2-inch)	102mm (4-inch)	152mm (6-inch)
Screened over a single interval (plastic screen and casing)			
1. Entire aquifer	$1,600-$3,700	$2,300-$4,500	$6,400-$7,500
2. Top 3 meters (10 feet) of aquifer	600- 1,050	700- 1,150	-
3. Top 1.5 meters (5 feet) of aquifer with drive point	100- 200	-	-
Piezometers (plastic screen & casing)			
1. Entire aquifer screened			
a. Cement grout	2,100- 4,700	2,800- 5,500	6,900- 8,500
b. Bentonite seal	1,850- 4,150	2,350- 4,950	6,650- 7,950
2. Top 3 meters (10 feet) of aquifer screened			
a. Cement grout	1,150- 2,050	1,200- 2,150	-
b. Bentonite seal	900- 1,500	950- 1,600	-
Well clusters			
1. Jet-percussion			
a. Five-well cluster, each well with a 6-meters (20-foot) long plastic screen	2,500- 3,800	-	-
b. Five-well cluster, each well with only a 1.5-meters (5-foot) long plastic screen	1,700- 2,300	-	-
2. Augers			
a. Five-well cluster, each well with a 6-meters (20-foot) long stainless steel wire-wrapped screen	4,600- 5,300	-	-
b. Five-well cluster, each well with only a 1.5-meters (5-foot) long gauze wrapped drive points	1,800- 2,600	-	-

TABLE 3 (Continued).

COST ESTIMATES FOR VARIOUS SAMPLING METHODS

Monitoring Technique & Construction Method	Price Per Installation Well Diameter 51mm (2-inch)	102mm (4-inch)	152mm (6-inch)
3. Cable tool			
a. Five-well cluster, each well with a 6-meters (20-foot) long stainless-stell, wire wrapped screen.	-	-	$ 9,850-14,150
4. Hydraulic rotary			
a. Five-well cluster, each well with a 6-meters (20-foot) long plastic screen, casing grouted in place.	-	$9,050-14,900	$13,800-19,400
b. Five-well cluster, completed in a single large-diameter borehole 4.5-meters (15-foot) long plastic screens, 1.5-meters (5-foot) seal between screens.	$4,240-5,880	$8,250-11,000	-
Single well/multiple sampling point.			
a. 33.5-meters (110-foot) deep well with 1-foot long screens separated by 1.2 meters (4 feet) of casing starting at 3 meters (10 feet) below ground surface.	-	-	$ 3,000- 4,700
Sampling during drilling	-	$3,000-4,700	3,300- 5,200

*Cost estimates are for an aquifer composed of unconsolidated sand with a depth to water of 3 meters (10 feet) and a total saturated thickness of 30 meters (100 feet). Cost estimates are based on rates prevailing in the Northeast in Autumn, 1975. Actual costs will be lower and higher depending upon conditions in other areas. Therefore, while the costs presented will become outdated with time, the relative cost relationships among the monitoring techniques should remain fairly constant.

- a pair of wells, one screened at the top of the other at the bottom of the aquifer;[20,21]

- a 3-well cluster with screens set on the top, middle and bottom of the aquifer under investigation,[22]

- clusters in which the screened intervals are separated by preselected intervals, such as:

    - the 3-, 6-, 9-, 12-, and 18-meter (10-, 20-, 30-, 40-, and 60-foot) screen depths;[23]

    - the 6-meter (20-foot) separation from 6 to 30 meters (20-100 feet) (Figure 33);[24]

    - terminating 2 to 3 wells at 3 to 4.5 meters (10-to-15-foot) intervals.[25]

The fixed sampling depth, whatever the screen placement selected, limits to some degree the effectiveness of the well cluster.

Some uncertainty will always exist as to the actual vertical distribution of the contaminant. Construction of more wells per cluster is not the answer; only a limited number of wells can be constructed close enough together to delineate vertical contaminant distribution at one particular point. Also, construction costs and the time required to complete the cluster would become prohibitive factors. The only way to obtain the most complete picture of leachate distribution is to collect ground-water samples during drilling. This procedure is described in section 3.2.5. However, this technique provides only a one-time series of samples.

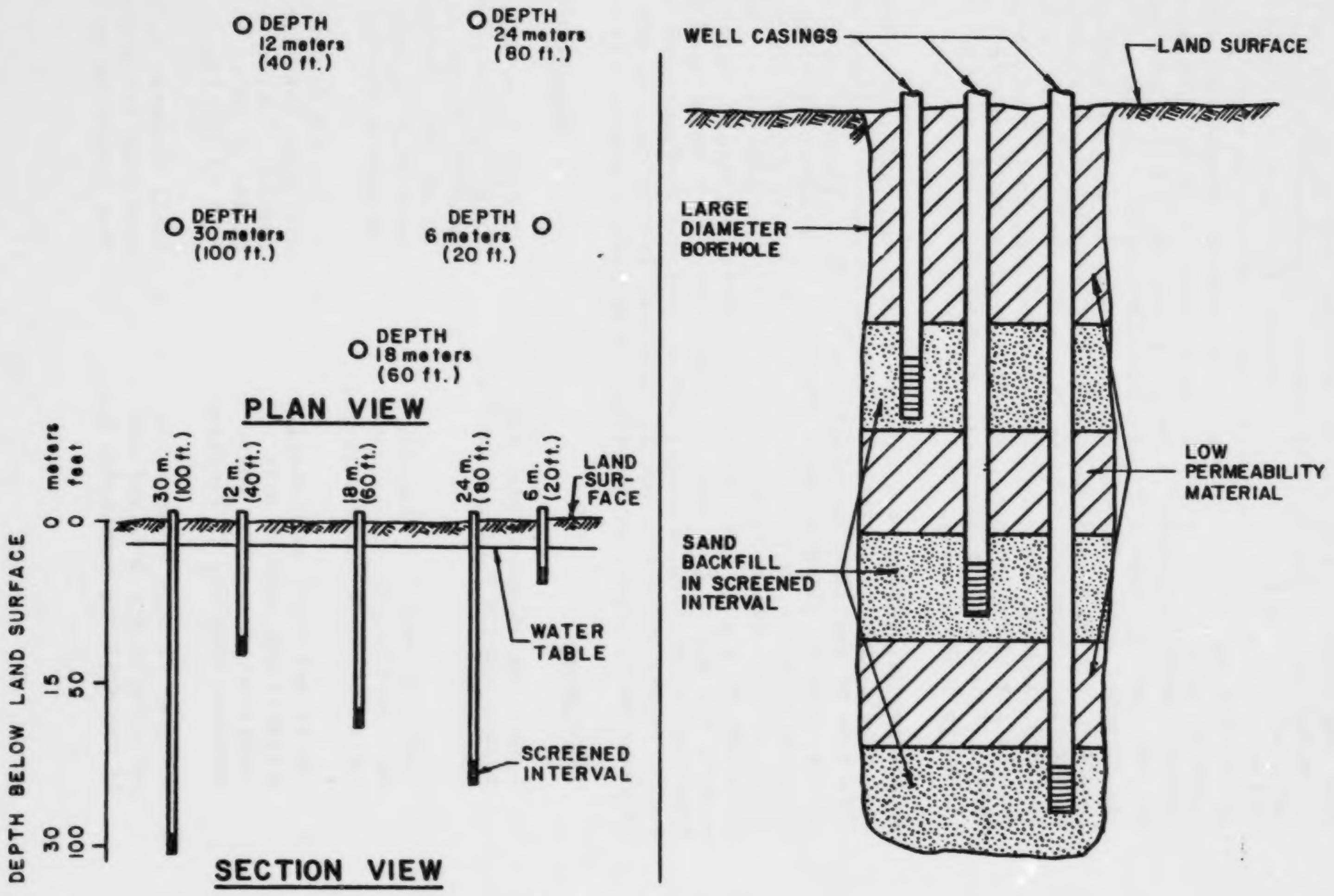
3.2.3.2 Implementation. Under certain conditions, well clusters are easily installed--a major factor to be considered when designing a leachate monitoring system. Normally, in unconsolidated sediments, a small-diameter steel casing 51 to 63mm (2 - 2½ inches) is driven by jet drilling to the desired depth; the screen is positioned by casing pull-back or by augering a hole and forcing a well point to the desired depth. Alternatively, a hole can be drilled or augered to a predetermined depth and a well point driven out the bottom of the hole into undisturbed sediments. Either installation technique is relatively rapid and inexpensive. For shallow aquifers (6-9 meters--20-30 feet), 32mm (1¼-inch) well points can be driven by hand to construct a cluster.[25]

Another approach to well cluster construction is multiple well completions in a single borehole. This involves drilling a large-diameter hole, using rotary technique or a bucket auger and installing small-diameter wells to selected depths with each screened zone isolated from the others by an impermeable seal.

Meyer (1973) completed as many as three 102-mm (4-inch) PVC wells in a single 559-mm (22-inch) borehole[27] and Huges et. al. (1971) installed up to six 32 to 51-mm (1¼-to-2-inch) observation wells in one boring.[28] This technique for constructing well clusters is feasible provided the cost of drilling large-diameter boreholes is not prohibitive and care is taken in placing the impermeable seals between screened zones (Figure 33). There is a great advantage in being able to construct the wells sufficiently close to obtain samples that are truly representative of a single point (areally) in the aquifer. This increases the value of the water-quality data obtained. If the seals between the individual wells are carefully constructed and precautionary measures are taken, such as using a shrinkage-inhibitor in the cement grout, reliable samples of ground water can be obtained. Of course, the greater the number of casings in the borehole, the greater the likelihood of imperfect seals between the casings. To insure that the seals are effective, a check on water levels in wells not being sampled should be made. An abrupt drop in water level would tend to indicate a vertical connection between the screens.

A variety of drilling methods can be used to install a well cluster in the hypothetical aquifer. One of the best methods is jet percussion; the seal between casing and formation is tight, and drilling charges are relatively low. This installation consists of a cluster of five wells (51mm--2-inch diameter) with screens from 3 to 9, 9 to 15, 15 to 21, 21 to 27, 27 to 33 meters (10 to 30, 30 to 50, 50 to 70, 70 to 90, 90 to 110 feet) below ground surface. The cost ranges from $2,500 to $3,800. Using screens only 15-meters (5-feet) long at depths of 9, 15, 21, 27, and 33 meters (30, 50, 70, 90, and 110 feet), the installation cost of the well cluster would range from $1,700 to $2,300. As previously discussed, however, the gaps between the screened intervals could allow a thin plume of leachate-enriched water to pass through undetected, rendering the latter type of construction less satisfactory.

Another way to construct well clusters in this aquifer is to use a power auger. In this method, sediment is loosened by a flight of augers, and a well point and 51-mm (2-inch) casing are pushed through the loosened material to the desired completion depth. This construction is problematic in that there is a potential for vertical leakage of water through the column of loosened soil around the well casing. Also, well screens must be sturdy enough to withstand the stress of being driven through the loosened sediments in the borehole. In normal practice, relatively inexpensive drive points are used (1.5 meters--5 feet--long or less). However, monitoring the entire saturated thickness of the hypothetical aquifer would require five 6-meters (20-foot) long, stainless-steel, wire-wound screens with a drive point. These would considerably increase the total cost of the installation. An augered five-well cluster with 1.5-meters (5-foot) long, inexpensive drive points should cost $1,800 to $2,600 while the most effective installation from a sampling standpoint with five


DEPTH
12 meters
(40 ft.)
DEPTH
24 meters
(80 ft.)
DEPTH
30 meters
(100 ft.)
DEPTH
6 meters
(20 ft.)
DEPTH
18 meters
(60 ft.)
PLAN VIEW
DEPTH BELOW LAND SURFACE
meters
feet
0
0
15
50
30
100
30 m. (100 ft.)
12 m. (40 ft.)
18 m. (60 ft.)
24 m. (80 ft.)
6 m. (20 ft.)
LAND SURFACE
WATER TABLE
SCREENED INTERVAL
SECTION VIEW
WELL CASINGS
LAND SURFACE
LARGE DIAMETER BOREHOLE
LOW PERMEABILITY MATERIAL
SAND BACKFILL IN SCREENED INTERVAL

(After Yare, 1975) 24

FIGURE 33. TYPICAL WELL CLUSTER CONFIGURATIONS

6-meters (20-foot) long, 51-mm (2-inch) diameter stainless-steel screens would cost $4,600 to $5,300--significantly more.

More expensive alternatives are drilling with either the cable tool or hydraulic rotary methods--the former costing about $9,850 to $14,150 per cluster of 152-mm (6-inch) diameter wells and the latter from $13,800 to $19,400. The substantial difference in these figures is due to the necessity of grouting the annulus between casing and borehole wall in the rotary-drilled holes. Grout is necessary to prevent vertical leakage of water through the annular material into the screen, causing samples to be unrepresentative of formation water in the screened zone. The presence of a seal between the casing and borehole wall in a cable-tool well renders grouting unnecessary. No substantial economies could be obtained by switching to 102-mm (4-inch) diameter cable-tool holes. Drilling costs would be about the same as for 152-mm (6-inch); the main savings would come from using 102-mm (4-inch) stainless-steel, wire-wound screens in lieu of 152-mm (6-inch) screens.

An alternative method to a cluster constructed with five individual well completions is to install multiple casings in a single borehole. This type of installation would cost $8,200 to $11,000 for five 102-mm (4-inch) diameter wells installed in a 610-mm (24-inch) diameter borehole, and in the range of $4,240 to $5,880 for five 51-mm (2-inch) wells installed in a 305-mm (12-inch) hole. The formation of a good seal between each screen is vital; therefore, screens 4.6 meters (15 feet) in length would have to be used in the hypothetical aquifer, allowing for a 1.5-meter (5-foot) seal between each screened interval. Again, this seal is critical; if not properly constructed, an anomalous water-quality sample will result.

Advantages

1. Simple installation does not always require hiring a drilling contractor.
2. Excellent vertical sampling made possible if sufficient number of wells are constructed.
3. "Tried and true" methodology, accepted and used in most contamination studies where vertical sampling is required.
4. Low cost if only a few wells per cluster are involved and if the drilling contractor has

Disadvantages

1. If only a few wells are installed, large vertical sections of the aquifer are unsampled. Artificial constraint on data by completion depths.
2. If jetting rigs or augers are used, installations are usually limited to total depths of 38 to 46 meters (125 to 150 feet).
3. Small diameter wells can be used only for monitoring. They cannot be used in

equipment suitable for installation of small-diameter wells.

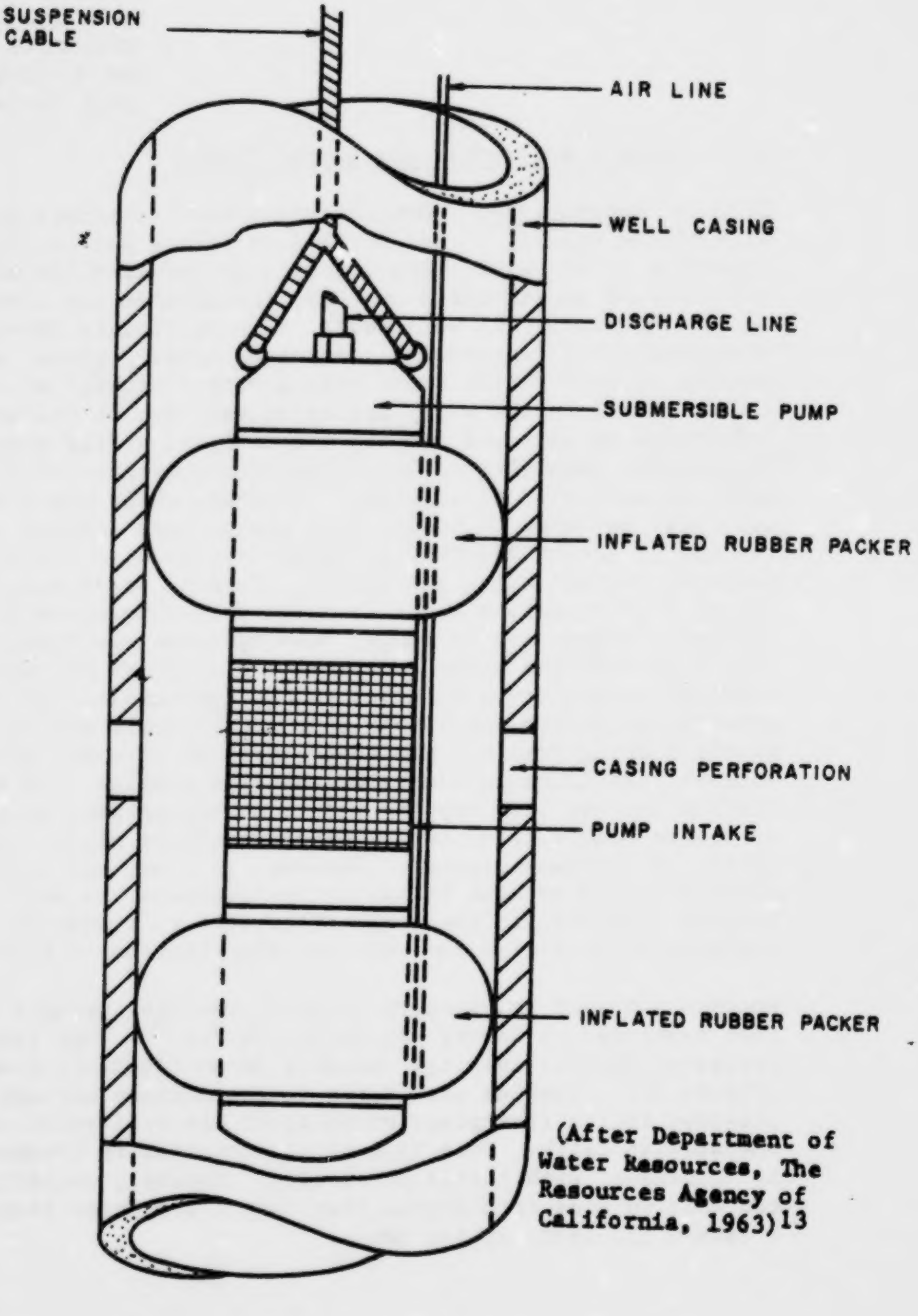
abatement schemes.

4. In small-diameter wells, development and sample collection become tedious and difficult if water level is below suction lift.

3.2.4 Single Well--Multiple Sample Points

3.2.4.1 Methodology. Another method used to sample multiple horizons in a single well is to set screens or casing perforations at regular intervals in the well. Spacing will depend upon the sample density required and construction expense; the greater the number of open zones, the higher the well costs. The California Department of Water Resources (1963) successfully obtained closely-spaced ground-water samples by perforating steel casing with a mechanical perforator at set intervals in the well, isolating each set of perforations with inflatable packers and pumping the isolated casing segment with a submersible pump (Figure 34).[13] The attractiveness of this type sampling operation is apparent. However, there are some pitfalls. Care must be taken to insure that the packers isolate the sampled section of screen and that no water from above or below leaks past the packers, contaminating the sample. Pumping rates must be kept low to insure that formation water is drawn only from opposite the screened section. Higher and prolonged pumping rates may induce significant flow from horizons above and below the level of the aquifer being sampled, resulting in an unrepresentative sample. If the annulus between the casing and borehole is not properly sealed, the influence of the pumping gradient may cause vertical movement of leachate in the annulus, resulting in non-representative samples. To adequately protect against this type of sample contamination, an impermeable seal of either bentonite or cement grout should be placed between every screen or slotted interval. However, this may not be possible with closely-spaced screens or casing perforations. A well constructed and sampled according to these specifications will provide an excellent opportunity to define the vertical distribution of a contaminant.

Another approach for shallow vertical sampling using a single well is that described by Hansen and Harris (1974).[29] They isolated fiberglass probes at regular spacings inside a 32-mm (1¼-inch) diameter well point (Figure 35). Samples were drawn to the surface through a tube attached to the fiberglass probe after the well point was driven to the desired depth. This type of construction is inexpensive and can be "homemade" with little difficulty. However, collection of samples can only be done from depths less than the suction limit--about 9 meters (30 feet) at sea level.


SUSPENSION CABLE
AIR LINE
WELL CASING
DISCHARGE LINE
SUBMERSIBLE PUMP
INFLATED RUBBER PACKER
CASING PERFORATION
PUMP INTAKE
INFLATED RUBBER PACKER

(After Department of Water Resources, The Resources Agency of California, 1963)13

FIGURE 34. USE OF A SAMPLING PUMP TO ISOLATE CASING PERFORATIONS

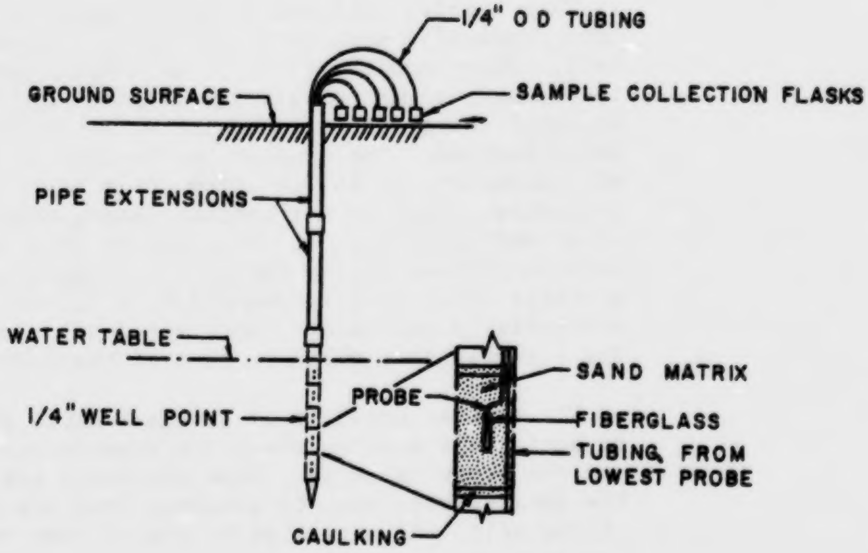

1/4" O D TUBING
GROUND SURFACE
SAMPLE COLLECTION FLASKS
PIPE EXTENSIONS
WATER TABLE
SAND MATRIX
PROBE
1/4" WELL POINT
FIBERGLASS
TUBING, FROM LOWEST PROBE
CAULKING

FIGURE 35. CONSTRUCTION DETAILS OF A GROUND-WATER PROFILE SAMPLER USING A WELL POINT

(After Hansen & Harris, 1974)[29]

3.2.4.2 Implementation. Unless the shallow well-point device as described by Hansen and Harris (1974) is used, installing a multiple sampling point well requires the services of a drilling contractor. A large-diameter borehole is needed in which casing can be positioned, necessitating the use of a cable tool or rotary rig. One hundred and fifty-two mm (6-inch) diameter or larger casing should be used to accommodate the packer-pump unit which in turn requires a tripod winch, power, and air supply. If steel casing is not slotted before installation, a special down-hole tool is required to make the slots.

The packer pump is not quite so formidable. Once a good-quality submersible pump has been obtained, local investigators or machine shop personnel can equip it with packers. Cherry (1965) has described the design and operation of a rather elaborate packer pump (Figure 36):[29]

> ...This sampler collects a pumped sample of water from a specific zone in an uncased or multi-screened well. Minor modification of the sampler permits remote measurement of several chemical and physical characteristics of the water in the zone being sampled. The sampler can be used in wells with diameters of 203 to 406mm (8-16 inches) inclusive, which do not contain pumps, pipes, or other obstructions. It is suspended on a cable from an A-frame and is raised and lowered by an electric motor that is powered by a 110-volt, a-c portable generator. This generator also runs the electric pump which is part of the sampler...(sic)
>
> ...The sampler consists of two inflatable packers or boots--one mounted above the submersible pump and the other below it. When the boots are inflated, the zone between them is isolated from the remainder of the well, and water can be pumped from this isolated zone... (sic)
>
> ...The capacity of the pump is about 1 l/s (15 gpm). The spacing of the boots can be varied by using different lengths of connecting pipe between them. The minimum spacing of the boots is 1.5 meters (5 feet) (length of pump). The boots are inflated by pumping water into them, from the well, through an electrically controlled valve; they are deflated by pumping the water out of them through another electrically controlled valve. Advantages of this sampler over other packer-type samplers are its portability, the ease with which it can be repositioned without removing it from the well, and the fact that it is relatively inexpensive...(sic)

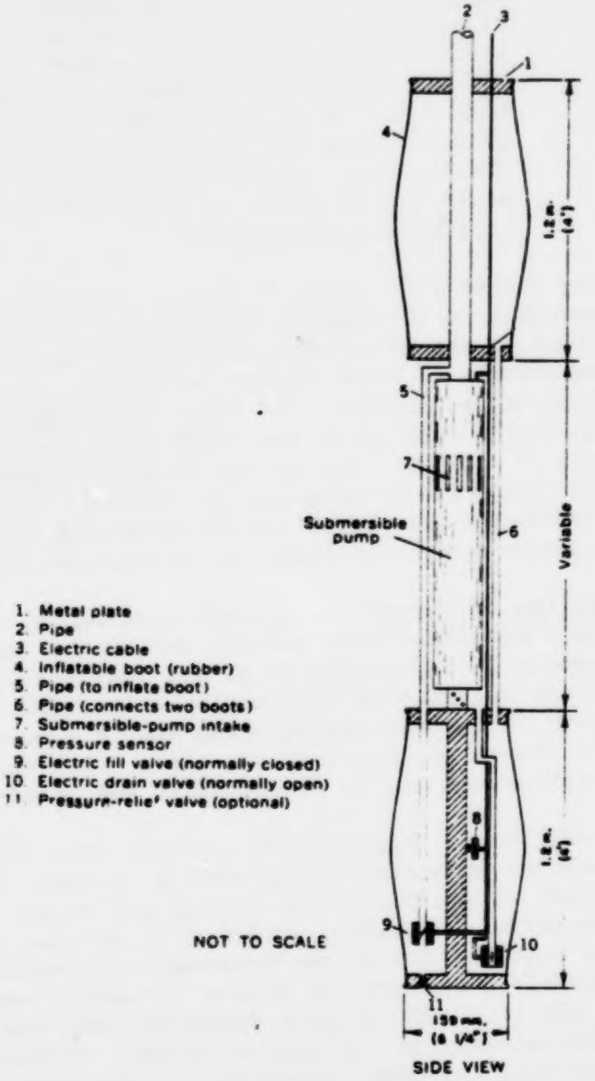

2
3
1
4
1.2 m.
(4')
5
7
Submersible pump
6
Variable
1. Metal plate
2. Pipe
3. Electric cable
4. Inflatable boot (rubber)
5. Pipe (to inflate boot)
6. Pipe (connects two boots)
7. Submersible-pump intake
8. Pressure sensor
9. Electric fill valve (normally closed)
10. Electric drain valve (normally open)
11. Pressure-relief valve (optional)
8
1.2 m.
(4')
NOT TO SCALE
9
10
11
159 mm.
(6 1/4")
SIDE VIEW

FIGURE 36. THE CASEE SAMPLER

(After Cherry, 1965)[30]

> ...Instruments to measure temperature, specific conductance, or other chemical or physical characteristics of the water in the well can be placed in the space between the boots. Experience has shown,that continous measurement of specific conductance, in this way, is very useful in determining the proper time to collect the water samples. It is desirable to pump from the well a volume of water equal to at least 3 times the capacity of the discharge line and isolated section before collecting samples for analysis, in order to ensure the collection of representative samples ...

The packer/pump shown in Figure 34 is less elaborate and more amenable to fabrication without the facilities of a machine shop. Although actual construction is not described in the California Department of Water Resources report, it seems that two rubber diaphragms, possibly cut from tire inner tubes, were clamped (probably with stainless-steel hose clamps) to the exterior of the pump.[13] An air line to the surface allows inflation or deflation of the packers. If sampling is limited to shallow depths, "homemade" diaphragms and valves should withstand the required inflation pressures, greatly reducing fabrication costs of the packer/pump.

The use of 152-mm (6-inch) PVC casing, slotted PVC screen, and glued-joint couplings will greatly facilitate construction of a well capable of being sampled at set intervals within the casing. The screen sections of set length can be separated by the appropriate lengths of blank casing using only a handsaw to cut the lengths and PVC cement to join them together. With the use of simple tools, this construction can be done rapidly and requires little skill.

A well in the hypothetical aquifer with .3-meters (1-foot) long screen sections separated by 1.2 meters (4 feet) of casing would cost \$3,000 to \$4,700. Steel casing and screen would be considerably more expensive. If casing perforation is done, its cost will depend upon the availability of the necessary equipment. The pump/packer assembly necessary for sample collection could be fabricated for \$1,000 to \$2,000 or more depending upon the pump used and the elaborateness of the packer system. Portable generators capable of supplying the power necessary for the pump can be purchased for several hundred dollars. Although these prices seem high, they are one-time costs; with proper care and maintenance, the pump/packer system should endure for years.

Advantages	Disadvantages
1. Excellent information is gained on vertical distribution of the contaminant.	1. Expensive.
2. If necessary, well diameter is large enough to use in a pumped-withdrawal pollution abatement problem.	2. Proper well construction and sampling procedures are critical to successful application.
3. Sampling depths are limited only by the size of the sampling pump.	3. Complicated sample collection procedure involves a great deal of equipment.
4. Rapid installation possible.	

3.2.5 Sampling During Drilling

3.2.5.1 Methodology. A major disadvantage of the previously-discussed sample collection techniques is that a constraint is placed on the data obtained from the ground-water samples by the fixed or arbitrary point at which water samples are collected, i.e. "blind" placement of wells can result in a false representation of contaminant distribution if it is not uniformly dispersed throughout the aquifer at the sampling point. Contaminants are often stratified: underlain, overlain, or interfingering with uncontaminated ground water. To define these relationships adequately, information on the vertical distribution of contaminant must be obtained prior to installation of the monitoring well. This information can be obtained by formation-water sampling during drilling.

Several researchers have obtained satisfactory ground-water samples during drilling using three basic techniques:

- driving a casing[31] or well point[32] to the desired depth, bailing or pumping a water sample from that depth, and repeating the process to completion depth or refusal;
- drilling a mud rotary hole to the sampling depth, pulling the drilling string, setting and gravel packing a temporary well screen, and pumping a formation water sample;[24]
- drilling a borehole to the desired horizon, setting a packer and riser pipe, and pumping a sample.[33]

As was true of the other techniques previously discussed, if proper precautions are taken, formation-water samples collected using these techniques will be representative of water quality at a known vertical interval of the aquifer. The critical factor in successful application is the development of the temporary well to the point where all traces of drilling fluid have disappeared from the pumped water before a sample is collected. Dilution of the sample by the drilling fluid and contributions of chemical constituents by clay particles in the mud will produce erroneous data, and little information will be gained on the actual vertical distribution of contamination.

The main advantage of the type of sampling is that the stratification of the contaminated slug can be defined with reasonable accuracy prior to setting a permanent casing and screen. With this information, the well can be designed for the most advantageous sampling or withdrawal of the contaminant at that point in the aquifer. Changes in the vertical distribution can then be monitored very closely.

3.2.5.2 Implementation. In the course of investigating a hexavalent chromium contamination problem, Yare (1975) used ground-water-sampling-while-drilling techniques.[24] A borehole was drilled using a hydraulic rotary rig. Use was made of a drilling mud containing an organic-base drilling fluid additive which minimizes the effect of drilling fluid on the formation water. A slotted PVC screen attached to a riser pipe was lowered to the bottom of the borehole; the screen zone was packed with fine gravel, forming a well in the borehole. After the gravel had settled through the drilling mud and around the screen, the well was pumped clear. By this time, an effective filter cake had formed on top of the gravel pack, isolating the screen from the fluid in the borehole. Due to the hydrostatic force created when pumping the well, the filter cake between the gravel and the borehole wall disintegrated. To insure that formation water was collected, an additional 379 liters (100 gallons) of water were pumped before sample collection. It should be noted that the use of an organic-base drilling fluid would be precluded if organic contaminants were to be determined.

The drilling and water-sampling procedure which evolved from this investigation consists of the following steps (Figure 37):

- drilling an 203-mm (8-inch) diameter borehole to the sampling horizon;
- pulling the drill string and replacing the bit with a 1.5-meter (5-foot) long, 102-mm (4-inch) diameter wire-wound well screen;
- lowering the screen and drill string to the bottom of the hole and gravel packing the screen;
- attaching a gasoline-powered centrifugal pump

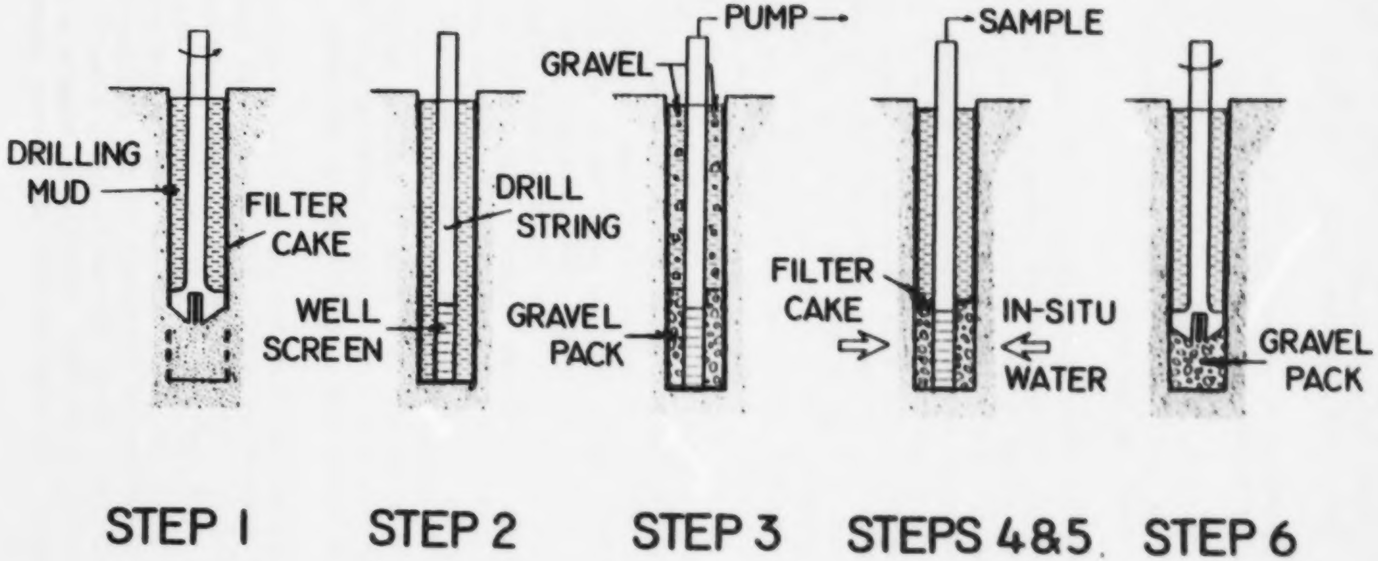

PUMP →
SAMPLE
GRAVEL
DRILLING MUD
FILTER CAKE
DRILL STRING
WELL SCREEN
GRAVEL PACK
FILTER CAKE
IN-SITU WATER
GRAVEL PACK
STEP 1
STEP 2
STEP 3
STEPS 4&5
STEP 6

FIGURE 37. IN-SITU GROUND WATER SAMPLING PROCEDURE

(Yare, 1975)[24]

to the drill string, pumping until the drilling-fluid level stabilizes in the hole, and the discharge clears of drilling fluid--(In this case, a centrifugal pump could be used as static water levels ranged from 1.8 to 3.6 meters (6-12 feet) below ground surface);

- pumping at least 379 liters (100 gallons) of formation water before collecting the samples;

- pulling and removing the screen, lowering the bit and drill string, and drilling to the next sampling horizon.

Harden (1974) described a sampling-during-drilling technique useful for deep holes in unconsolidated sediments. In this method, the first hole drilled is 171mm (6 3/4 inches) in diameter. When the hole penetrates approximately 4.6 to 9.2 meters (15-30 feet) into a sand from which a water sample is desired, drilling is stopped (step 1, Figure 38). The hole is reamed to a diameter of 251mm (9 7/8 inches) down to a point just above the zone selected for water-sample collection. At this point, the 171-mm (6 3/4-inch) hole is washed out to its original depth (step 2, Figure 38); and a string of pipe with packer and screen is positioned in the hole (step 3, Figure 38). The pipe is usually 102mm (4 inches) in diameter; the packer is a commercial rubber-cone type with typical dimensions of 152 by 229 by 356mm (6x9x14 inches). Often a canvas "shirt tail" is wrapped by the packer to assist sealing. The packer is positioned on the shoulder at the junction of the 171-mm (6 3/4-inch) and the 251-mm (9 7/8-inch) portions of the hole. Below the packer, a commercial 102-mm (4-inch) well screen 3 to 6 meters (10-20 feet) long is attached to the 102-mm (4-inch) pipe. After the packer is seated, the temporary well is developed by airlift for several hours until the water becomes clear. The air line is removed from the 102-mm (4-inch) pipe; a small-diameter pump is installed; and the temporary well is pumped once more until the water becomes clear. The samples are then collected. When pumping comes to a halt, the casing and screen are pulled from the hole; and drilling of the 171-mm (6 3/4-inch hole) is resumed until a second water-bearing zone is encountered from which a water sample is desired. At this time, the entire water-sampling process is repeated.

In the case of the hypothetical aquifer previously described, the most efficient sample-collection technique would probably be that described by Yare (1975). If this method is used, samples can be obtained at depths of 6, 9, 12, 15, 18, 21, 24, 27, 30, and 33 meters (20, 30, 40, 50, 60, 70, 80, 90, 100, and 110 feet) below ground surface. Five of the most contaminated intervals of the aquifer can be screened with 1.5 meters (5-foot) long, 102-mm (4-inch) plastic screens for a total cost of $3,000 to $4,700. Additional sample points would cost $125 to $200 each. The samples from the screen segments could be collected

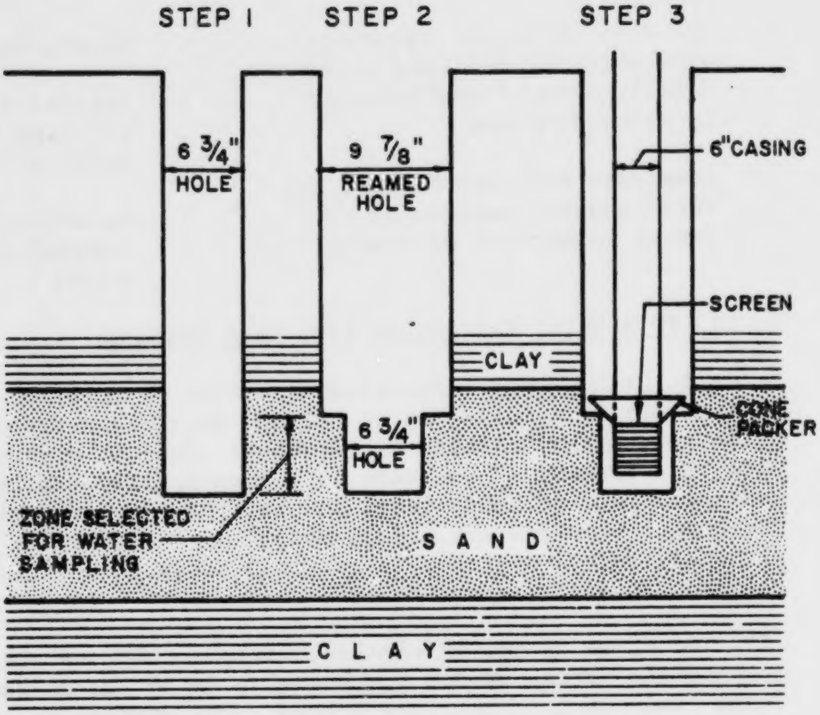

STEP 1
STEP 2
STEP 3
6 3/4"
HOLE
9 7/8"
REAMED
HOLE
6"CASING
SCREEN
CLAY
CONE
PACKER
6 3/4"
HOLE
ZONE SELECTED
FOR WATER
SAMPLING
SAND
CLAY

FIGURE 38. PROCEDURE FOR WATER SAMPLING DURING DRILLING

(Harden, Personal Communication, 1974)[33] by using a packer/pump or by installing and isolating a deep pressure-vacuum lysimeter in each screened interval. If a packer/pump is used, 152-mm (6-inch) casing is necessary; the total cost per installation would be in the range of $3,300 to $5,200. Lysimeters could be placed for approximately $100 each.

Advantages

1. The best technique currently available for defining vertical distribution of contaminants in thick aquifers.
2. Completed well can be used for water-quality monitoring and/or pumped withdrawal of contaminant.

Disadvantages

1. Considerably expensive.
2. Careful supervision of drilling and sampling is necessary.
3. Potential cross-contamination of samples exists.

3.2.6 Pore-Water Extraction From Core Samples

3.2.6.1 Methodology. An indirect method of obtaining ground-water samples during drilling is to extract pore water from core samples collected as the borehole is advanced. This is accomplished by placing a segment of the core in a commercially available filter press, hydraulic ram, or centrifuge, and forcing interstitial water out of the core. Standard analytical techniques can then be used on the extracted water, typically electrical conductivity measurements and titrametric analysis for chloride.[34, 35, 36]

This technique is not infallable, however. During the process of driving the coring device and bringing it to the surface, drilling-fluid invasion into the core occurs. The greater the invasion, the less reliable the water-quality data obtained. Sand and gravels are more readily invaded than finer-grained sediments.

Luscynski (1961) overcame this problem by putting fluorescein dye (green color) in the drilling mud.[34] Any penetration of drilling mud into a core sample would be indicated by the dye. The uninvaded core sections could be selected for extraction. In this way, dilution of pore water by drilling fluids would not be a factor, and a representative sample could be assured. Unfortunately, disposal of the bright-green drilling mud upon completion of drilling becomes a problem.[37] Normally, drilling mud is dumped on the ground and is eventually eroded away. Because of its natural gray or brown color, it is not very obvious. The fluorescein in the bright-green mud, although not believed toxic in dilute concentrations, is visible at very low concentrations in water. If left at the site, its migration to nearby wells could cause complaints from well owners.

3.2.6.2 Implementation. Luscynski (1961) describes in detail the use of a filter press to extract pore water:[34]

> ...The filter press used is one of several types sold commercially for use in determining filtration properties of drilling muds. The unit consists of a chamber, a filtering medium, a graduated tube for catching and measuring the filtrate, and a pressure-source unit. A cell, base cap, screen, rubber gaskets, and top cap make up the chamber. The cell is 89-mm(3½-inches) high and has inside and outside diameters of 76 and 89mm (3 and 3½ inches), respectively. The filtering medium is a sheet of filter paper which fits on the screen over the base cap; it has a filtering area of about 45cm^2 (7 square inches). The filter paper used is specially hardened to withstand the pressure in the chamber. A graduated cylinder is used to catch the filtrate... (sic)
>
> ...Uninvaded cores consisting of loose material such as sand and gravel are transferred to the filter-press chamber by spatula or spoon. Usually it is not practical to remove much more than 25% to 50% of the uninvaded material from the core barrel by this method. Enough material is transferred to fill the chamber about a quarter to half full. It is then tamped lightly until an integrated unit is formed in the chamber and a film of water is formed on the surface of the material and along the cylindrical wall...
>
> ...Usually less than 10 to 20 percent of a solid or salty clay sample is invaded by the drilling fluid. Plugs of the uninvaded material 25.4 to 51mm (1-2 inches) long are placed in the chamber to fill it about a quarter full. Then they are molded and tamped into an integrated unit. Difficulty can be experienced in molding and ramping a relatively dry solid clay--one having a water content of less than 15 to 20 percent of the dry weight of the clay... (sic)
>
> ...The total time between the opening of the split-spoon core barrel and the placing of the chamber in the frame is usually 60 to 120 seconds for the loose material and 120 to 300 seconds for the tight material. There is thus, very little opportunity for evaporation...

> ...After a sample is properly prepared for filtration, the chamber is fully assembled, placed in the frame, and made airtight by the T-screw (Figure 39). The gas pressure is then applied.
>
> ...The pressure of the carbon dioxide gas in the chamber moves some of the interstitial water through the filter screen and filter tube into the graduated cylinder. Pressures of 34.5 to 207 kN/m^2 (5-30 psi) suffice for the gravel, sand, and silt samples. Pressures of about 690 kN/m^2 (100 psi) are usually sufficient for silty and solid clay samples. The carbon dioxide gas does not alter the chloride concentration of water forced from the material into the filtrate tube... (sic)
>
> ...Chloride determinations of the filtrate are made in the field by the standard titration method using silver nitrate solution. Relatively large amounts of filtrate (25 to 50 ml) are needed when fresh water is to be tested, and relatively small amounts (1 -10 ml) when salt water or diffused water is to be tested...
>
> ...Usually enough filtrate can be obtained from the uncontaminated material of only one core if it is taken in a diffused-water or salt-water zone. However, more than one core may be necessary to obtain the required amount of filtrate if the core is taken in a fresh-water zone; this is particularly true for solid-clay samples which yield only small amounts of interstitial water...

An alternative to the mud filter press is to fabricate a core sample squeezer which utilizes a hydraulic ram, as described by Manheim (1965):[38]

> ...The squeezer utilizes a commercially available cylinder and ram (made by the Carver, Co., Summit, N.J.) to which a machined base with a filtering element and fluid outlet is fitted. Construction details are shown in Figure 40. The filter unit consists of a stainless steel screen and a perforated steel plate contained in a circular recess in the steel filter holder. Alternatively, a porous (sintered) metal plate may be used to replace both screen and perforated plate. The top surface of the filter should be flush with the other rim of the holder so that it may support one or more paper filter disks. Finegrained, hardened laboratory filters give a visually clear effluent, but

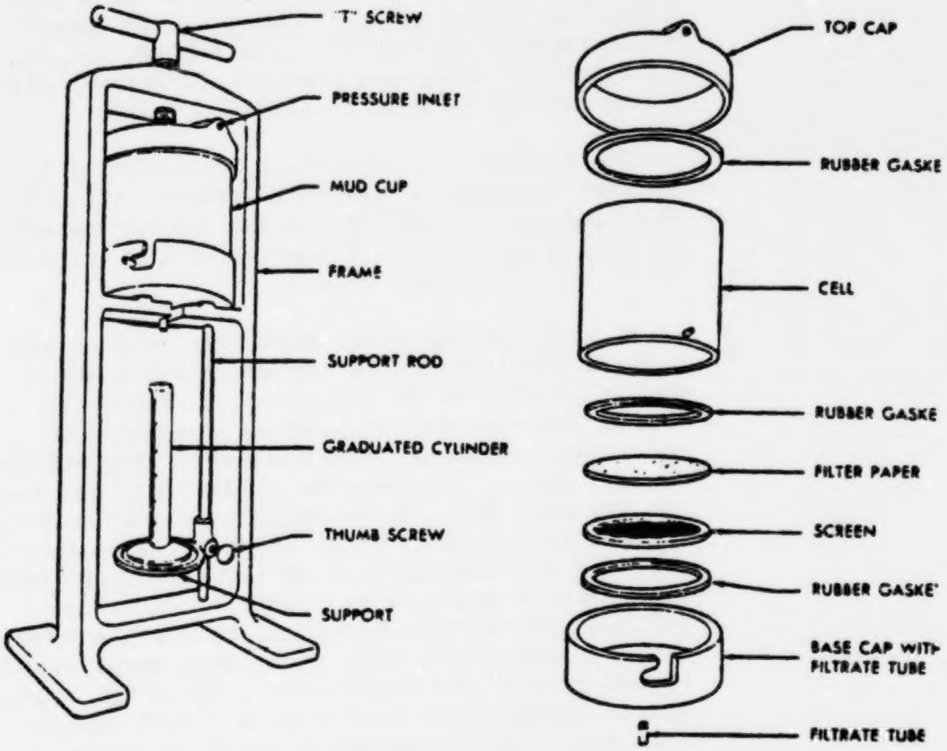

"T" SCREW
PRESSURE INLET
MUD CUP
FRAME
SUPPORT ROD
GRADUATED CYLINDER
THUMB SCREW
SUPPORT
TOP CAP
RUBBER GASKE
CELL
RUBBER GASKE
FILTER PAPER
SCREEN
RUBBER GASKE'
BASE CAP WITH
FILTRATE TUBE
FILTRATE TUBE

FIGURE 39. FILTER PRESS

(As manufactured by Baroid Division, NL Industries,
Houston, Texas)

membrane or microfilters may be used to assure maximum freedom from suspended matter. The lower part of the filtering assembly, fitted with a rubber washer, protrudes into a recess in the steel base; when pressure is applied, the gasket is squeezed against the cylinder and prevents leakage of water around the filter unit. The space in the recess and the diameter of the outflow boring are kept small so that little fluid can collect in the squeezer itself. All metal parts in the filter base are made of Iron and Steel Institute No. 303 stainless steel. Rubber and teflon disks just below the piston prevent loss of fluid upward when pressure is applied. The rubber, teflon, and filter-paper disks are punched out with an arbor punch and can be made as needed...(sic)

...The present design permits insertion of a disposable syringe (preferably plastic) directly into the base of the squeezer to receive fluid. The narrow effluent hole is reamed out to permit fitting of the standard 'Luer' taper of the syringe nose...

...A larger squeezer has also been constructed using a 64-mm (2½-inch) Carver cylinder and piston. The design is similar to that shown in Figure 39 except that the filter plate is increased in thickness to give greater strength. The effluent line remains small. Because the cross-sectional area of the cylinder bore of the small squeezer is about 22-mm (0.88-inch), a 10.16-tonne (10-ton) laboratory press exerting its maximum load of 9,090kg (20,000 pounds) will apply a pressure of about 151,724 kN/m^2 (22,000 psi) to the sediment. However, a 9,090-kg (20,000-pound) load will apply only about 34,482 kN/m^2 (5,000 psi) in the large unit. The large squeezer should therefore be used with a higher capacity press when more compact sediments are to be squeezed...(sic)

...In sequence, the steps in squeezing a sample are as follows: The filter holder with its gasket is placed in the recess of the filter base. The screen, perforated plate (or porous disk) and 2 or 3 filter-paper disks are positioned. The cylinder is seated over the filter unit so that it rests firmly on the base. Sediment is then quickly transferred into the cylinder through the top, followed by the teflon and rubber disks. The teflon and rubber disks can be placed above the sample in

> either order to obtain a leak-free pressure transfer, but placing the teflon disk below the rubber disk gives a cleaner seal than the reverse order shown in Figure 40. The piston is depressed as far as it will go into the cylinder, and the whole unit put in the press for squeezing. Pressure is applied gradually at first; when the first drop of interstitial fluid is seen, the syringe is seated in its hole in the base (effluent passage in Figure 40). The squeezed-out liquid moves the plunger of the syringe back as the liquid is expelled, and there is minimum opportunity for evaporation. When the desired amount of liquid has been obtained, the syringe is removed and capped...(sic)
>
> ...After extraction of the liquid the parts of the apparatus are rinsed with distilled water and (except for rubber parts) with acetone. The acetone helps dry the unit quickly in preparation for the next sample. The squeezing and washing operations together can be completed in 300 to 600 seconds (5 to 10 minutes)...(sic).

The main expenditure in this type of sampling is the filter press. The current price for this piece of equipment can be obtained from Baroid Division, NL Industries, Houston, Texas. Current charges for cores obtained by wire-line, 51-mm (2-inch) diameter split-barrel samplers are $30 to $50 per core. A section of core can be taken from the sampler, molded into the filter press, the fluid extracted and analyzed for chloride concentration, and measured for specific conductance in a half hour or less.

Advantages	Disadvantages
1. Generally inexpensive.	1. Quantitative analysis requires careful control during sample collection.
2. Pore water extract is amenable to field chemical analyses such as: chloride concentration and specific conductivity.	2. Interstitial water can drain from unconsolidated sand and gravel reducing volume of the collected water sample.
3. Excellent vertical sampling when mud invasion into core sample is monitored.	3. If dyed drilling mud is used, it may be an eyesore.
4. Samples can be obtained from almost any depth when wire-line coring apparatus is used.	4. Core recovery in coarse sand and gravel can be difficult and time consuming.

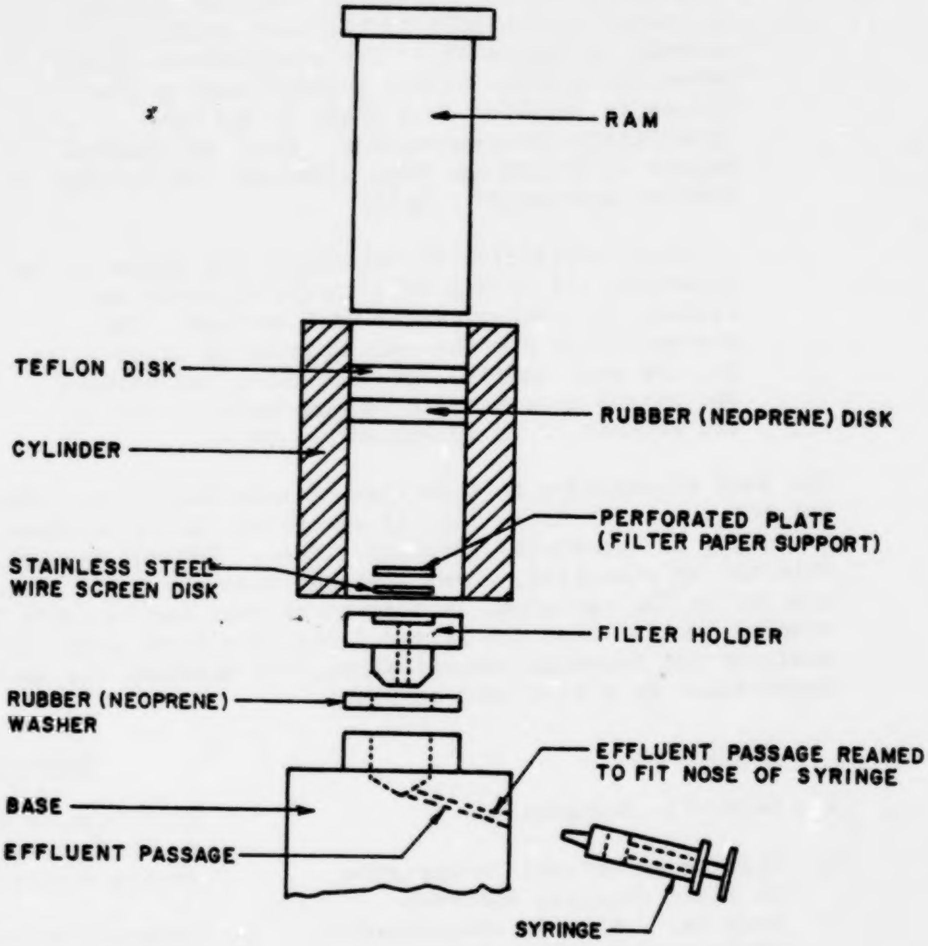

RAM
TEFLON DISK
RUBBER (NEOPRENE) DISK
CYLINDER
PERFORATED PLATE
(FILTER PAPER SUPPORT)
STAINLESS STEEL
WIRE SCREEN DISK
FILTER HOLDER
RUBBER (NEOPRENE)
WASHER
EFFLUENT PASSAGE REAMED
TO FIT NOSE OF SYRINGE
BASE
EFFLUENT PASSAGE
SYRINGE

FIGURE 40. HYDRAULIC SQUEEZER

(After Manheim, 1966) 38

5. Qualitative use of pore water extract allows for presence/ absence determination.	5. Small sample volume available for chemical analysis.
6. Can be used with consolidated rock as well as unconsolidated sediment samples.	6. Can be expensive.

3.2.7 Summary of Cost Estimates

Table 3 presents a summary of cost estimates for the various monitoring techniques used in the zone of saturation. The cost estimates are based upon the hypothetical aquifer which was supposed to be composed of unconsolidated sand with a depth to water of 3 meters (10 feet) and a total saturated thickness of 30 meters (100 feet). It should be realized that these cost estimates are based upon prevailing rates (Fall, 1975) in the Northeast; consequently, actual costs will be lower or higher, depending upon conditions in other areas. Also to be considered are the rising costs of drilling and materials over the past few years. Therefore, the costs presented here will soon be outdated. However, these estimates indicate the relative cost relationships that should remain fairly constant.

3.2.8 Selection of Well Size

The casing size required for a monitoring well is principally dependent upon the particular sample-withdrawal technique to be utilized (see Chapter 6). There is no justification establishing a blanket well-size requirement resulting in additional cost especially when any cost savings can be reinvested to establish additional monitoring points and a more comprehensive program.

Where the ground-water sampling point is within suction lift (usually less than 9 meters--30 feet), vacuum or pressure sampling methods can be used with small diameter wells (51mm--2inch). When sampling beyond suction lift, bailers or submersible pumps are used which would normally require a large well diameter (102mm--4-inches). A pressure sampling method can also be used, however, which would only require a 51-mm (2-inch) well.

The type of soils should also be considered. For example, in extremely tight soils where recovery is slow (hours or days), a large diameter may be desired so that one casing volume provides an adequate sample quantity. Also, if the well will serve as a ground-water pumping well to control contamination, this will affect the well size.

3.3 FIELD INSPECTION

Field inspection is an extremely valuable tool in evaluating landfill sites. Although an inspection by a trained observer would produce more data, even an unskilled person can identify the presence of leachate in springs, seeps, and streams by its color and odor. Frequently, vegetation that has been exposed to leachate can be found in a dead or dying state. The condition, surface configuration, and drainage away from a landfill give insight into the amount of infiltration of precipitation that might be taking place. A study of surface drainage, topography, and nearby wells enables the inspector to make an estimate of ground-water (and leachate) movement. Field observations increase in value when combined with geohydrologic information and other pertinent basic data contained in published reports and agency files.

Many of the monitoring techniques discussed in this chapter can be combined with the field inspection to provide even more information. The degree of success depends upon the ability of the inspector to interpret the situation and the amount of time available for the study.

The inspector should have a detailed map or aerial photograph of the landfill site (or at least a sketch map). While touring the landfill and the surrounding acreage, the findings should be recorded on the map, giving an overall picture that can be easily interpreted.

The cost of a landfill field inspection is variable according to the operation size and the complexity of the surrounding terrain. In this and all the following sections of this chapter, the probable expenditures involved are estimated. Costs are based upon an estimated daily wage rate for the required personnel, the approximate time required for the various tasks (based upon an average situation), and other related expenses such as laboratory fees and expense accounts. For a field inspection of an average landfill (20.2 hectores--50 acres), one hydrogeologist or engineer (or equivalent) would be required for 2 to 3 days at $200 per day. The estimated cost is $600.

Advantages	Disadvantages
1. Can be carried out quickly and inexpensively.	1. Untrained inspector may overlook subtle but valuable data.
2. Helps place the overall problem in perspective.	2. Findings are not always conclusive in detecting ground-water contamination.
3. Establishes the extent of additional investigations which may be required.	3. Time factors are not indicated relative to condition changes.

4. When combined with a literature survey on available data, inspection procedure may be used by an experienced hydrologist to roughly establish the overall situation.	4. Few, if any, analyses or actual physical measurements are made.
5. Provides an opportunity for personal communication with landfill operator and other personnel.	5. Untrained inspector may be misled by visually impressive but environmentally insignificant features.

3.3.1 Seeps

Small springs of discolored, malodorous leachate which are frequently found along the lower edges of many landfills are referred to as seeps. These may be the only visible indication of landfill leachate and thus often receive more than their share of attention. In fact, they may represent only a very small fraction of the total leachate being generated by the landfill. The few gallons per minute visible in seeps are insignificant when compared with the possible hundreds or even thousands of liters (gallons) of unseen leachate which may be migrating downward to the water table. However, as indicators of leachate, seeps do deserve consideration.

Seeps may represent the intersection of the water table and the land surface, or they may be the discharge from a small perched water body within a landfill. At times, a distinction between these two situations can be made by inspection. For example, if the land surrounding the landfill is dry, a seep discharging along the face of the solid waste is not likely to represent the water table. Installation of a well point near the landfill would establish the true water-table position near the seep and provide a more definite distinction between the two situations. Such well points have the added advantage of permitting a sample of ground water to be collected and tested for leachate.

Seeps are valuable sources for the collection of concentrated leachate samples--although it should be kept in mind that it is possible that the seep may not always be representative of the large volume of leachate generated in that particular landfill. In fact, the chemical characteristics of any leachate sample, regardless of its source, should not be considered representative of the total volume of leachate. Landfill leachate has proven to be highly variable in relation to location and the period of time. When there are substantial changes in seep locations or flow rates or the sudden appearance of new ones, a change in the flow system within the landfill is indicated. The exact nature and cause of the change, however, must be investigated by other means.

Examination of seep would typically be included as a part of the field inspection and, except for laboratory fees for analyses, would not represent an additional expense.

Advantages	Disadvantages
1. Where present, definite indication of leachate generation.	1. May not indicate presence of contaminated ground water.
2. Convenient point of collection for leachate sample.	2. Chemical quality not necessarily representative of bulk of leachate in the landfill or entering the ground water.
3. Changes in flow rates or locations of seeps are indicative of interval landfill changes.	

3.3.2 Vegetation Stress

Leachate contamination may result in stress and possible destruction of vegetation on the surrounding area. Stressed species may include agricultural crops, stands of trees, and marsh or meadow vegetation.

In marsh environments subject to leachate discharge, the condition of vegetation serves as an excellent monitoring device to assess ecological stress on the total system. Marsh vegetation is stationary and sensitive and can be studied for signs of stress, either by using aerial remote-sensing techniques or direct study by the field botanist. Crops and trees generally grow in areas with a deeper water table than is associated with the marsh environment and are more likely to be stressed by landfill-generated gases than by leachate. Various types of agricultural crops, including fruit orchards, have been destroyed by migrating gases generated within a nearby landfill. Preliminary stresses placed upon these species prior to their actual destruction are often detectable by the botanist and by aerial remote sensing.

While identifying the precise cause and mechanisms of stress can be prohibitively costly, it may be possible to relate the stress to a general cause which may in turn be related to the presence of the landfill. Mapping the extent of stressed vegetation may provide an indication of the extent of the total impact of a landfill on its surrounding environment. Also, early detection of stress may permit the institution of corrective measures in sufficient time to prevent irreparable damage.

A cursory investigation of vegetation stress would be included in the field inspection and would not represent an additional expense. A detailed survey of vegetation stress (including an assessment of probable cause) would require 1 to 2 days of field work by a botanist. Some laboratory work would also be involved. The estimated cost of such a survey is $1,000.

If vegetation stress indications are to be used for monitoring or if specific recommendations regarding the saving or replacement of stressed species are to be made, the required program might cost between $10,000 and $100,000, depending upon the extent, complexity, and goals of the program.

Advantages	Disadvantages
1. Qualitative indicator of leachate and gas contamination.	1. Evidence of stressed vegetation, especially in early stages, is not always evident except to a trained botanist.
2. Mapping extent of stressed vegetation may provide an indication of the limits and source of contamination.	2. Stress may be caused by many factors, some unrelated to the presence of the landfill. Determination of the responsible factor or factors is usually extemely difficult.
3. Stressed vegetation can be mapped remotely by aerial photographic methods, allowing wide coverage in a short period of time.	3. Certain stresses will not occur unless physical or chemical change occurs at the surface or within the vadose zone. Therefore, it provides no indication of problems at depth.
4. Stress change is a good indicator for monitoring purposes. More effective if selected species are planted, then observed.	

3.3.3 Specific Conductance and Temperature Probes

Two physical characteristics of ground water which can be readily measured in the field are specific conductance and temperature. Since landfill leachate generally has substantially higher temperature and specific conductance than natural ground water, the presence of leachate often can be detected by these two characteristics.

Typically, *in-situ* measurements of ground-water characteristics would be made by lowering a remote-sensing probe into a well and recording the results from surface instrumentation. In areas of high-water table, however, the measurements can be made without installing a well. The method involves the use of a self-contained conductance-temperature probe. Construction details of such a device are shown in Figure 41.

The probe can be pushed directly into soft ground; where the ground is harder, it can be inserted into a small-diameter, hand-augered hole. During insertion, the perforations are protected from clogging by an outside tube. When the probe is below the water table, the tube is retracted, allowing the ground water to flow into it. Specific

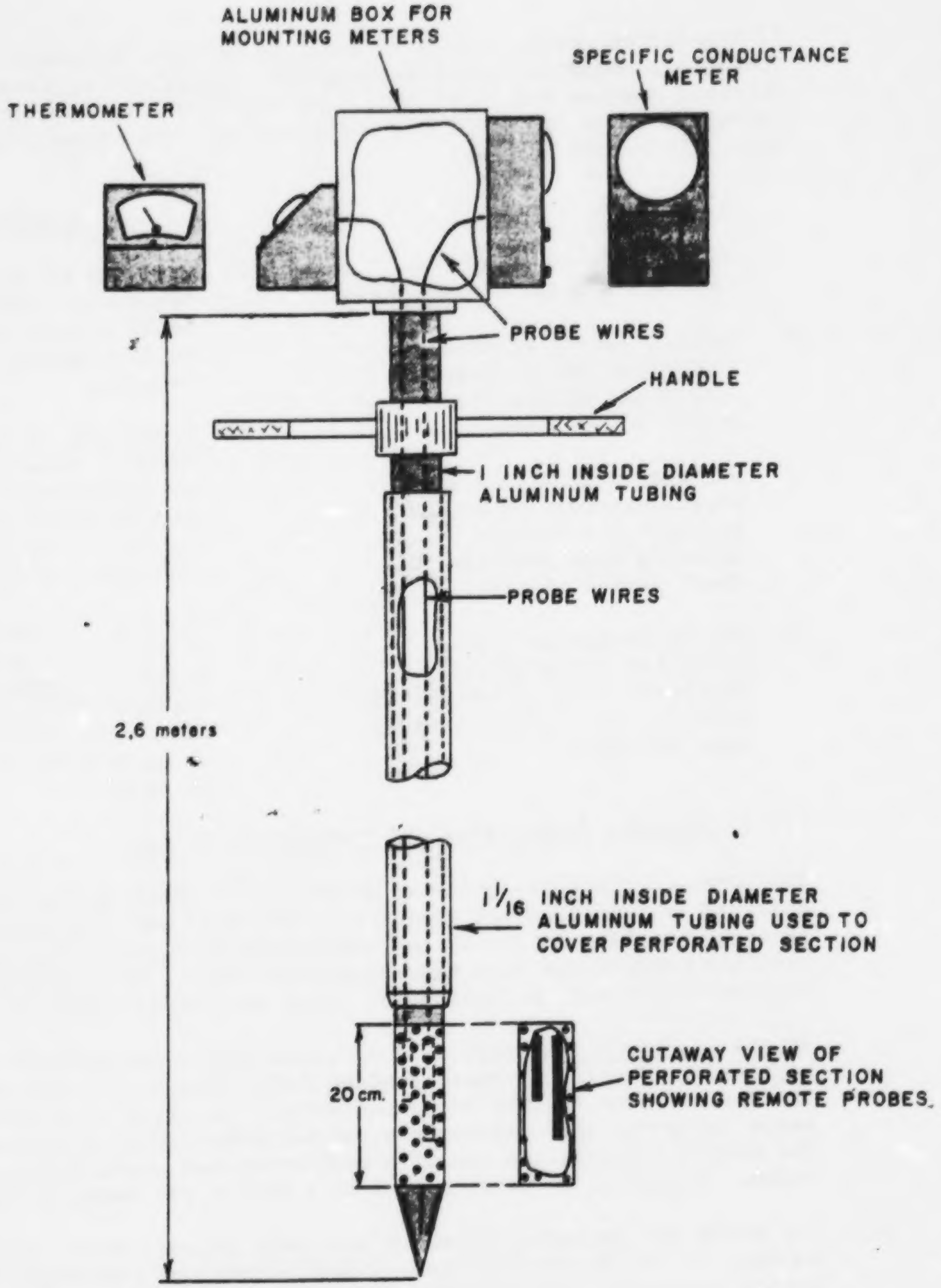

ALUMINUM BOX FOR MOUNTING METERS
SPECIFIC CONDUCTANCE METER
THERMOMETER
PROBE WIRES
HANDLE
1 INCH INSIDE DIAMETER ALUMINUM TUBING
PROBE WIRES
2,6 meters
1 1/16 INCH INSIDE DIAMETER ALUMINUM TUBING USED TO COVER PERFORATED SECTION
20 cm.
CUTAWAY VIEW OF PERFORATED SECTION SHOWING REMOTE PROBES.

FIGURE 41. SPECIFIC CONDUCTANCE - TEMPERATURE PROBE

conductance and temperature of the ground water can then be recorded. After removal, the perforated end of the probe is washed in clean water.

Another advantage of the specific conductance and temperature probe method is inherent in the equipment required. Since the mechanisms involved are not bulky or cumbersome, a two-man crew can easily carry all necessary equipment into the field and make a series of probe measurements over a typical landfill in 2 to 3 days. Also, swampy areas not readily accessible to drilling rigs or resistivity survey crews can be tested with little difficulty.

Although this method has distinct advantages, it is not an absolute. The required equipment for probing is very delicate and is vulnerable to physical abuse. Any malfunctioning of the equipment due to mechanical failure or to pre-contamination before testing, can give erroneous information. To prevent this development, the equipment should be checked periodically for malfunctioning against a standard solution such as potassium chloride.

Assuming the surface conditions were favorable for probe insertion, the cost of a ground-water conductance and temperature survey using the type probe discussed would be about $900. This estimate is based upon the cost of a hydrogeologist or equivalent and a helper over a period of three days. The survey may require more or less time depending upon the size of the site to be investigated and its accessibility.

Advantages	Disadvantages
1. Providing equipment is properly calibrated and insertion procedures carefully implemented, positive determination as to presence and degree of contamination can be made.	1. Not an absolute method. Equipment subject to malfunctioning, causing erroneous information. Equipment must be checked for malfunctioning against a standard solution.
2. Provides accessibility to otherwise restricted areas, such as marsh or swamplands.	2. Requires hiring personnel trained in the use and handling of the equipment.

3.3.4 Electrical Earth Resistivity

An electrical earth resistivity survey can be used to define subsurface geology and the extent of leachate contamination of ground water. The results of a resistivity survey, coupled with a minimal amount of direct sampling, may provide a basis for decisions on the necessity of remedial action or serve as a preliminary investigation from which a detailed drilling and sampling program is designed. However, resistivity is an indirect method and is subject to possible error in interpretation; final conclusions should not be based upon resistivity results alone.

The earth resistivity method depends upon the conduction of electric current through the subsurface materials. The magnitude and distribution of the current flow is a function of the effective resistivity (or its reciprocal, conductivity) of the subsurface material. Since the vast majority of the constituent minerals are poor electrical conductors, the effective resistivity of saturated materials is dependent upon moisture in interstices and pores. The pore spaces that contain water also contain some dissolved salts; it is these ionic solutions that allow the passage of current from the surface into the underlying material. It has been found that materials such as moist clays and silts have low resistivity; while in dry, loose soils, sand and gravel, or sand and gravel saturated with high-quality water, the resistivity is high. The electrical resistivity of a material is a function of the actual resistance of the material and the length of the current flow. Earth materials are not homogeneous; therefore, the measured resistivity is actually termed apparent resistivity and is defined as the weighted average of the actual resistivities of the individual subsurface materials or strata within the depth of penetration of the resistivity measurement.

To measure earth resistivity, a known current is introduced into the earth through two current electrodes. The resulting potential drop is measured between a second pair of potential electrodes. If the electrodes are arranged in a straight line and the separations are increased at constant increments, it is possible to make inferences about the relations of variations in apparent resistivity, depth of penetration, and electrode spacing.

Various procedures have been developed to interpret resistivity data. The procedures are grouped into two basic types:

- theoretical
- empirical.

Following the theoretical method, the field data are graphed and the resulting curve compared with sets of master curves developed for numbers of resistivity layers with definite ratios of resistivity and thickness. By using this method, the value of resistivity for each geologic unit as well as its thickness and depth can be determined. The example given on Figure 42 illustrates the empirical method. This procedure involves plotting the apparent resistivity and accumulated apparent resistivity values on a graph. The first curve indicates the type of material; the second shows the depth of the interface between layers.

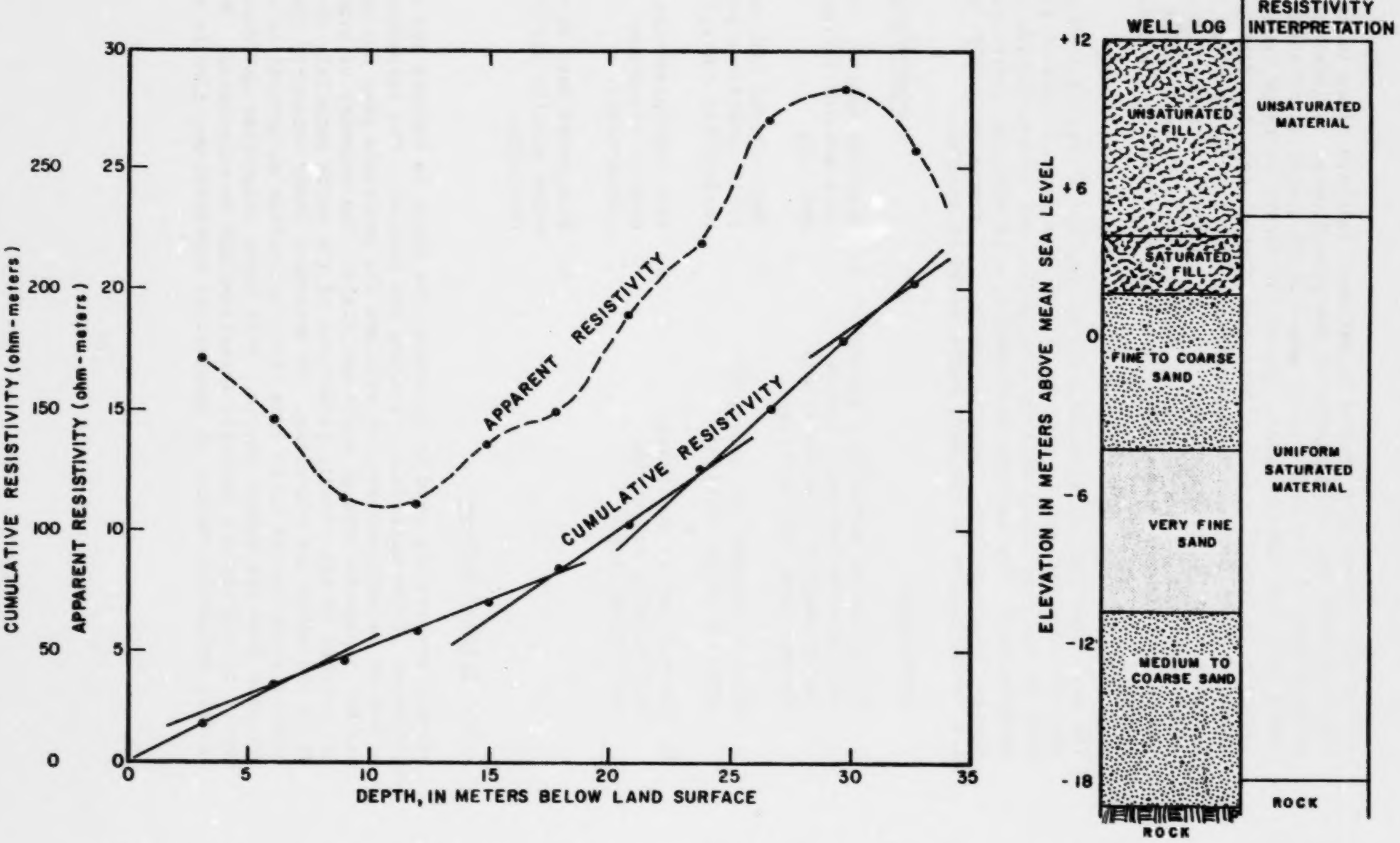


FIGURE 42. INTERPRETATION OF RESISTIVITY MEASUREMENTS USING THE EMPIRICAL CUMULATIVE METHOD

Use of the resistivity method to define a leachate plume relies upon the fact that the conductivity of the ground water is inversely proportional to the resistivity measured in a section of earth containing that ground water. Since the conductivity of landfill leachate is generally much higher than that of natural fresh ground water, a sharp decrease in apparent resistivity will occur if leachate is present in the measured section. By running a series of resistivity depth soundings on a grid over a landfill site, it is possible to define the lateral extent of the leachate plume by contouring the values obtained.

The estimated cost of a resistivity survey for a typical landfill (20.2 hectares ~50 acres) site is $1,000 based upon 2 days of field work by the crew along with data reduction and interpretation by a geophysicist. For surveys encompassing substantially larger areas, charges would increase proportionately. If access were difficult, i.e. brush had to be cleared, costs would be greater.

Advantages	Disadvantages
1. Definition of subsurface geology and contaminated water bodies can be derived at a faster and cheaper rate than drilling.	1. Indirect method. Requires some substantiation by drilling.
2. Greatly reduces the number of sampling wells required.	2. Many natural and man-made field conditions preclude resistivity surveys.
3. Surveys can be duplicated periodically to provide monitoring data.	3. Data interpretation in complex situations is often questionable.
	4. Background data on natural-water quality are prerequisite.

3.3.5 Seismic Surveys

Seismic surveys are used to determine the depth to bedrock and the thickness of the materials overlying the bedrock. The refraction method of seismic exploration utilizes the principle that energy waves can be propagated through earth materials. The velocity of propagation is governed by the elastic properties of the earth materials through which the waves are traveling. To determine their velocity, these elastic waves can be timed from their initiation to arrival at a known distance from the energy source. With known velocities and distances, depths to the various geologic interfaces can be calculated. The seismic reflection method of geophysical surveying may also be used.

This system, in which the energy wave is reflected from the different geologic horizons, can usually penetrate greater depths than the refraction method. It is not generally applicable to landfill studies. For more refined interpretations, well data are correlated with the results of the seismic survey. Where well information is not available, evaluation of seismic data is based upon interpretation of the geologic environment and experience in geophysics.

In using the refraction method to determine depths and seismic velocities of various materials, the reverse profile method is used. A reverse profile is defined as the interchange of the most distant energy source and the receiving unit (geophone); a second profile is then recorded. The energy source is a hammer blow on a steel plate or an explosive charge. With a single geophone seismic unit, a seismic profile is conducted by implanting the geophone firmly into the ground and moving the impact point away from the geophone at measured distances. In a multi-geophone unit, the geophones are placed at selected linear intervals; a single energy source, usually an explosive device, is activated. By recording the energy arrivals for different separations between the impact point and the receiver or receivers, a curve can be plotted, correlating energy travel-time with distance.

Data interpretation of a seismic survey requires a trained operator and an experienced geophysicist. The complexity of the data-reduction process generally requires the use of a computer. For these reasons, seismic surveys should be contracted to a firm providing geophysical services.

Though advantageous, seismic surveys are not without limitations. The seismic method cannot provide information on the presence or absence of leachate. For this reason, it cannot be considered a monitoring method in itself. Unlike resistivity surveys, repetition is pointless unless it is believed that the data are incorrect or insufficient.

The cost of a seismic survey would be essentially equal to a resistivity survey, i.e. $1,800 for a typical small landfill site.

Advantages	Disadvantages
1. Can provide subsurface geologic information much faster and cheaper than drilling.	1. Provides no direct information about leachate.
2. Can be used to extend geologic data over broad areas on a limited budget.	2. Requires more direct substantiation such as drilling.
3. Can be used in certain areas where access for a drilling rig would be difficult.	3. In complex geologic formations, interpretation is difficult and substantial errors may occur.

4. Requires the hiring of a trained person and the use of a computer to reduce and interpret data.

5. Subject to noise interference in many field situations.

3.4 OTHER MONITORING TECHNIQUES

3.4.1 Surface-Water Quality Measurements

Surface-water bodies, such as ponds or streams, in close proximity to landfills often have an orange color and an oily film on the surface. These obviously polluted water bodies are usually discharge points for contaminated ground water which originated within the landfill. Location of these discharge points on a topographic map of the landfill site will often help to provide a reasonable preliminary picture of the ground-water flow patterns. Where surface-water bodies are large or rapidly flowing, dilution of leachate as it discharges is often sufficient to prevent detection by visual inspection. In such cases, water samples should be taken and analyzed to establish the presence of typical leachate constituents, although even here dilution may interfere.

Prior to the collection of surface-water samples, *in-situ* measurements should be made by performing a survey for:

- specific conductance,
- pH,
- Eh,
- dissolved oxygen.

Such a survey can provide much useful information in itself; it will at least indicate the locations from which surface-water samples should be taken. The importance of an analysis of surface-water quality at a landfill site is twofold:

- determination of leachate discharge areas is crucial to establishing an overall hydrogeologic picture;
- surface-water quality degradation is an important component of overall environmental degradation and should be carefully examined.

Also, in a full investigation of surface-water bodies near a landfill, the native biota should be studied for the effects of leachate.

The estimated cost of a surface-water quality survey as described is $300, exclusive of laboratory fees. It is assumed that this program would be part of a more extensive investigation and that analysis of the results of the surface-water quality survey would be covered under a more general data analysis phase.

Advantages	Disadvantages
1. Useful in locating leachate discharge points.	1. Surface water may be subject to contamination from other sources not defined.
2. Can be a quick and inexpensive means of estimating environmental impact of the landfill.	2. Dilution may be too great to provide useful information.

3.4.2 Aerial Photography

Aerial photography has several important uses in landfill studies. Regardless of tone (black and white or color), an aerial photograph in its simplest form will show the landfill and drainage away from it. For large areas, remote sensing of vegetation stress by the use of aerial photography may be a justifiable undertaking. Advanced stress may be visible on color photography; less advanced stress may be distinguished by using infrared photography.

Another method which has been used in landfill investigations is the taking of multispectral aerial photographs. This procedures involves the use of special equipment to determine subtle differences in light reflected at various wave lengths for stressed and unstressed species. Photographic filters which emphasize this difference are used, and several images of the same area are made simultaneously with the aid of a multi-lens camera and the selected filters. Differences between stressed and unstressed vegetation are further enhanced by projecting the images through different color filters and superimposing them on a projector screen.

The usefulness of aerial photography is not limited to the detection of vegetation stress. Accurate contour maps of the landfill surface may be constructed from aerial photographs and are important in determining hydrologic characteristics of the landfill. As the surface changes, stereo color photography is used to construct and update topographic maps of the active landfill sites.

It is important that bench marks, wells, and other sampling points be located on the map. The inclusion of these items will facilitate problem interpretation.

Aerial photographs of a landfill site may be readily available from a local firm. If not available, it may be necessary to have the site photographed. Available photographs generally cost about $10 to $30 each. A series of black and white photographs generally costs approximately $100 to $300 for an overflight. Special photography such as color infrared or multispectral photography along with the necessary interpretation will cost up to $2,000 for a 20.2-hectores (50-acre) landfill site. This sum would include a topographic map and a map showing vegetation stress. A report of the result of the photo interpretation also would be included.

Advantages

1. Frequently can detect stressed vegetation which indicates contamination.
2. Can be used to prepare contour maps relatively inexpensively. Also provides certain geologic information.
3. Much less costly than a detailed ground survey of vegetation stress.
4. Yearly photographs can provide unbiased and indisputable evidence of surface changes such as: landfill configuration, vegetation conditions, and surface-water body locations.
5. Can be used to precisely map key wells and sampling points of the landfill site.
6. Enables a quick familiarization of the landfill site conditions without visiting the site.

Disadvantages

1. Availability of aerial photographs and photographic services is sometimes limited.
2. Little information concerning sub-surface conditions.
3. Little indication as to precise causes of detected surface changes.

3.4.3 Geophysical Well Logging

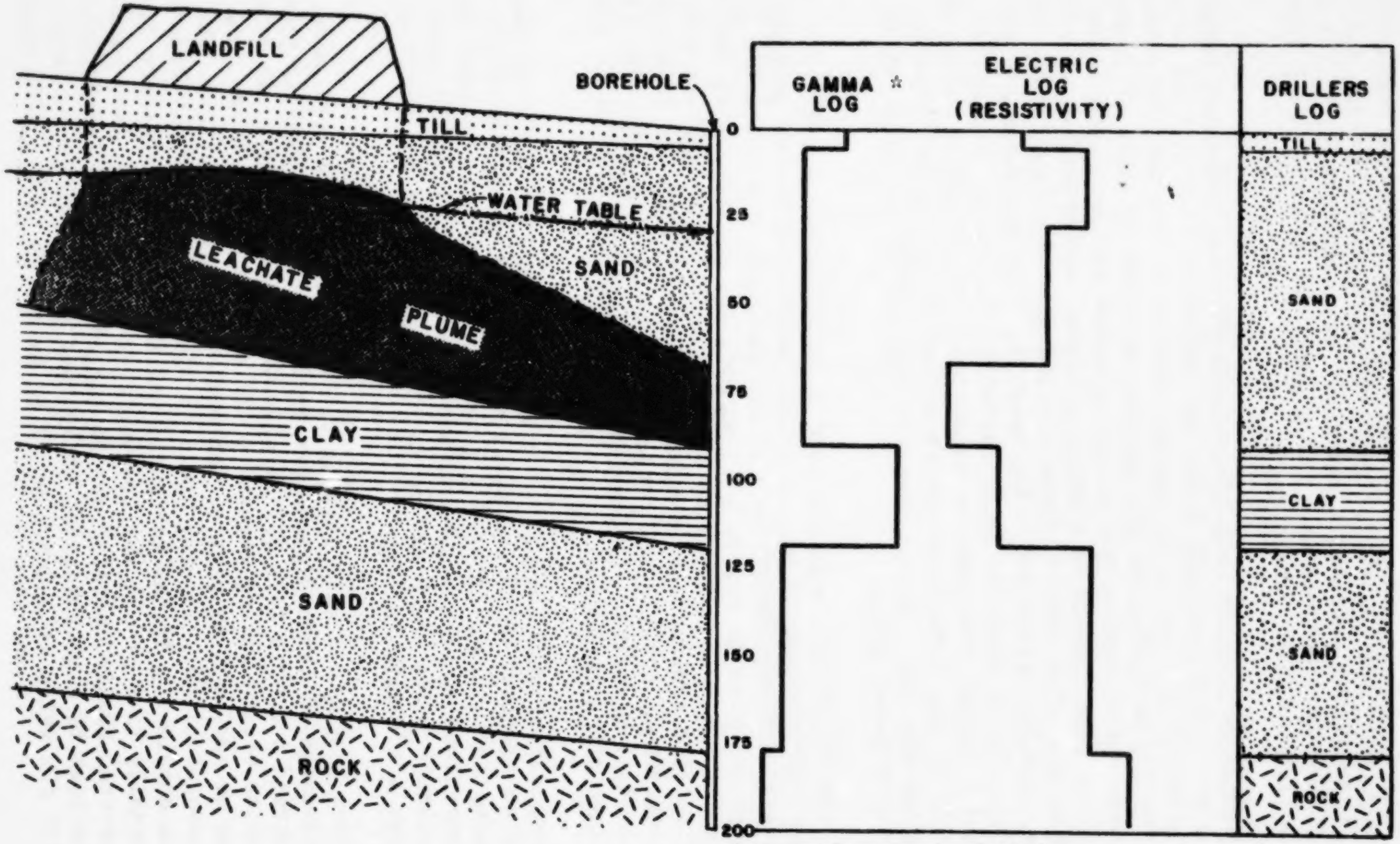
Geophysical well logging provides indirect evidence of sub-surface formations, indicating the relative permeabilities as well as the depths of the formations. The most common borehole geophysical operation is electric logging. This procedure consists of recording the apparent resistivities of the sub-surface formations and the spontaneous potentials generated in the borehole. This information is plotted against depth below the ground surface. The measurements of apparent resistivity and spontaneous potential are related to the electrical conductivity of the sediments--a partial function of the size of the grains. Thus, fine-grained sediments containing silt and clay will have a lower resistivity than clean, coarse sand and gravel. In addition, a leachate plume may be detectable by an electric log as illustrated schematically in Figure 43. Electric well logs can be run only in uncased boreholes.

Gamma-ray logging is a borehole geophysical procedure based upon measuring the natural gamma-ray radiation from certain radioactive elements that occur in varying amounts in sub-surface formations. The log is a diagram showing the relative emission of gamma-rays, measured in counts per second, plotted against depth below land surface. Since some formations contain a higher concentration of radioactive elements than others, formation changes with depth often can be accurately determined. For example, clay and shale contain more radioactive elements (e.g. isotopes of uranium, potassium, phosphorous, and thorium) than sand or sandstone. The relative amount of silt and clay in the formations can be estimated by the deflections of the gamma-ray log. Unlike electric logs, gamma-ray logs can be run in single-cased wells.

Geophysical well logs are used to supplement the driller's and geologist's logs of the materials penetrated by the borehole. An example of the comparison between geologic, electric, and gamma-ray logs is shown in Figure 44. An accurate evaluation of the sub-surface geology at a landfill site is essential to the determination of the direction and rate of movement of leachate from the landfill, and the contaminant attenuation capacity of the materials through which the leachate migrates. Geophysical well logs are helpful in evaluating these characteristics.

Geophysical well logging generally is applicable only to those landfill investigations which include test drilling and is therefore not an independent tool. However, gamma-ray logging can be used to gain some understanding of the sub-surface geology at a landfill site from existing wells which may be in the vicinity and for which no geologic logs are available.

Since geological well logging requires specialized equipment and the knowledge of trained operators to implement, the task is normally carried out by a firm offering geophysical services. In some instances, larger well-drilling companies are so equipped in which case the logging can be included as part of the well-drilling operation. The cost of geophysical well logging would be $300 to $500 per day depending upon the complexity of the equipment and size of the necessary crew. Normally five or six shallow wells or two to three deep wells (several hundred meters) can be logged in a day. Interpretation of the logs by a geophysicist would cost about $400 for a landfill site with six wells at a depth of 30 to 60 meters (100 - 200 feet).


LANDFILL
TILL
BOREHOLE
WATER TABLE
SAND
LEACHATE
PLUME
CLAY
SAND
ROCK
GAMMA LOG *
ELECTRIC LOG (RESISTIVITY)
DRILLERS LOG
0
25
50
75
100
125
150
175
200
TILL
SAND
CLAY
SAND
ROCK

*gamma log is included because it indicates that the leachate plume is not actually a clay layer.

FIGURE 43. DETECTION OF A LEACHATE PLUME USING AN ELECTRIC WELL LOG

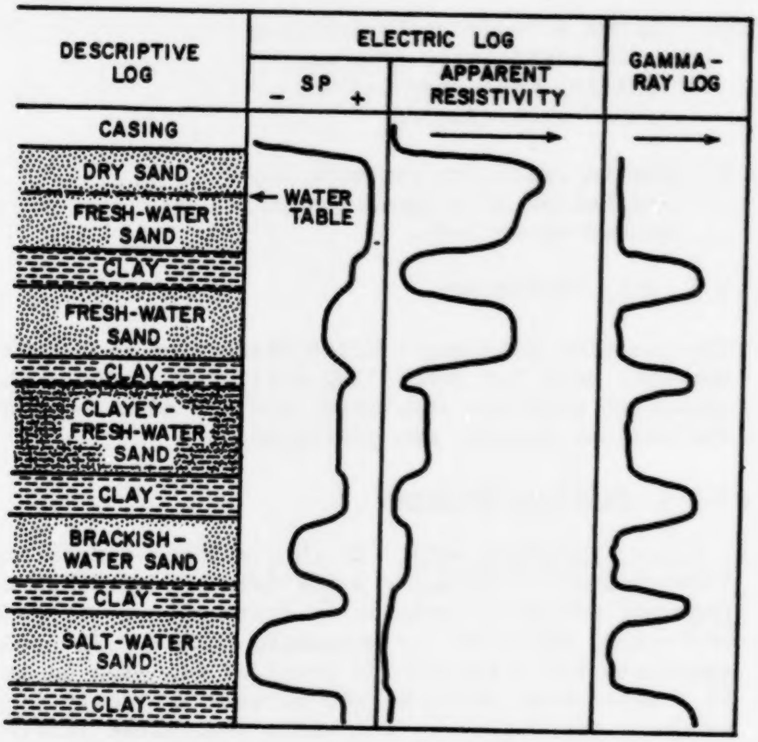

DESCRIPTIVE LOG
ELECTRIC LOG
SP
−
+
APPARENT RESISTIVITY
GAMMA-RAY LOG
CASING
DRY SAND
WATER TABLE
FRESH-WATER SAND
CLAY
FRESH-WATER SAND
CLAY
CLAYEY-FRESH-WATER SAND
CLAY
BRACKISH-WATER SAND
CLAY
SALT-WATER SAND
CLAY

FIGURE 44. COMPARISON OF GEOLOGIC, ELECTRIC, AND GAMMA-RAY LOGS

(After Johnson Division, UOP, 1972)

Advantages	Disadvantages
1. Provides back-up data to substantiate driller's and geologist's log of borehole.	1. Requires special equipment and the hiring of trained operators; thus, adding considerable expense.
2. Allows a more accurate determination of depth to formation change than might be achieved with routine sampling.	2. Is not an absolute for quantitative hydrogeologic determinations.
3. Allows a rough geological log to be constructed from an existing well that was not logged when drilled.	
4. May be useful in locating top and bottom of a contaminated ground-water body.	

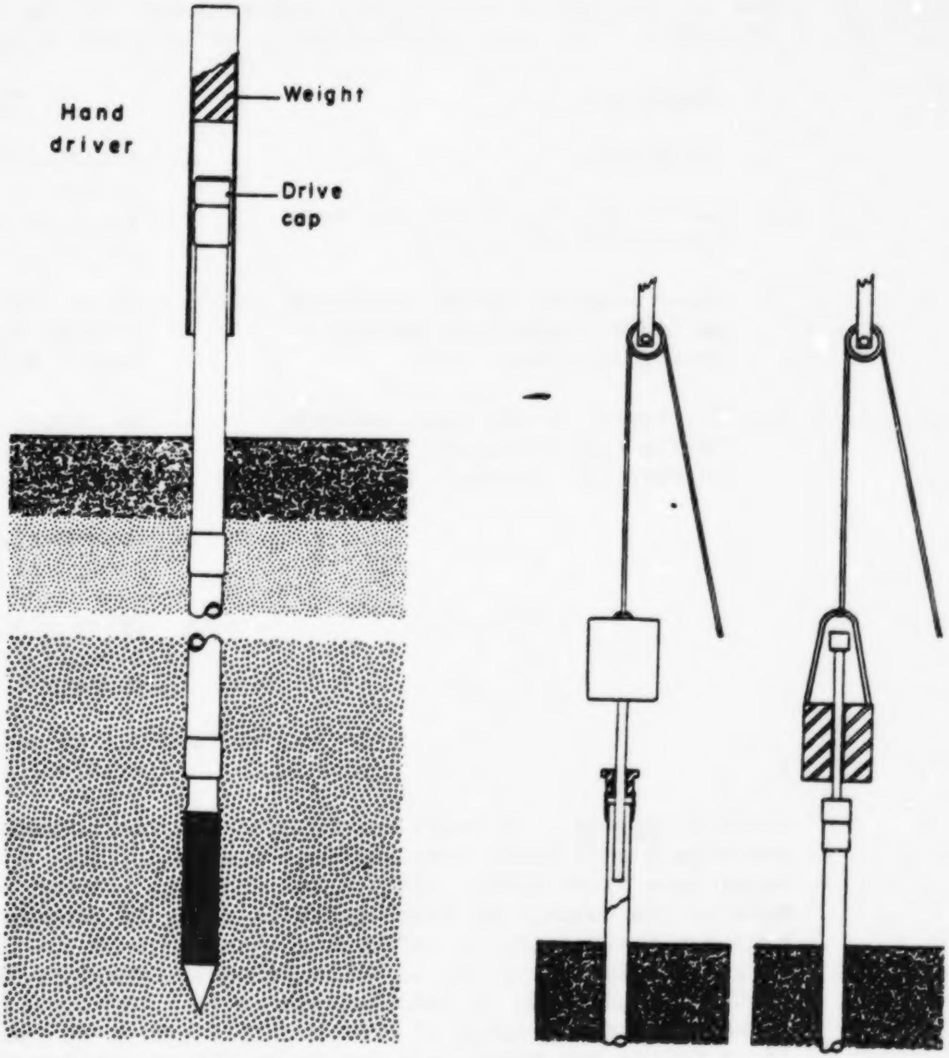
3.5 WELL TECHNOLOGY

This section presents a brief discussion of the various drilling methods commonly used for installing monitoring wells. The drilling method and equipment used are described, and the advantages and disadvantages of the various methods are presented.

3.5.1 Drilling Methods

3.5.1.1 Drive Points. In this method of drilling, a 31.7- to 51-mm (1¼- or 2-inch) diameter drive point is attached to a 51-mm (2-inch) pipe and driven to completion depth with a sledge hammer, drive weight, mechanical vibrator, or pneumatic hammer. The point can be driven to approximately 9 meters (30 feet) by hand and up to 30 meters (100 feet) if a mechanical drive weight is used (but only if driving is done in sands or finer-grained sediments that offer little resistance to penetration). Boulders cannot be overcome. Powell et al. (1973) report using a mechanical vibrator to drive points to depths of 20 meters (65 feet).[39] Drive points, because of their small diameter, are used in areas of high-water table from which water can be removed by suction pumps (e.g., pitcher pumps or centrifugal pumps).

Reliance on a drilling contractor to install drive points may not be necessary. Investigators can drive them with a minimal investment in equipment and manpower. The first step is to bore a vertical hole as deeply as possible with a hand auger slightly larger than the well point (Figure 45). The drive point is attached to a length of riser pipe (1.5-meter--5-foot lengths are preferable) and placed in the augered hole. A drive cap is placed on the top of the casing prior to driving.


Hand driver
Weight
Drive cap

HAND ASSEMBLY

HEAVIER ASSEMBLIES OPERATED BY DRILLING RIG OR TACKLE

Figure 45. Methods for Installing Well Points

(After Johnson Division, UOP, 1972)

Casing can be driven with a tool similar to the type used for driving steel fence posts or by a drive weight suspended from a tripod or derrick. Drilling will be more efficient if there is a source of power to lift the weight, as they can weigh from 34 to 204 kilograms (75 to 450 pounds). Drive points can also be driven with a sledge hammer, but this procedure is difficult and tedious. As the casing and point are driven, they are turned slightly to keep the threaded joint tight.

Advantages

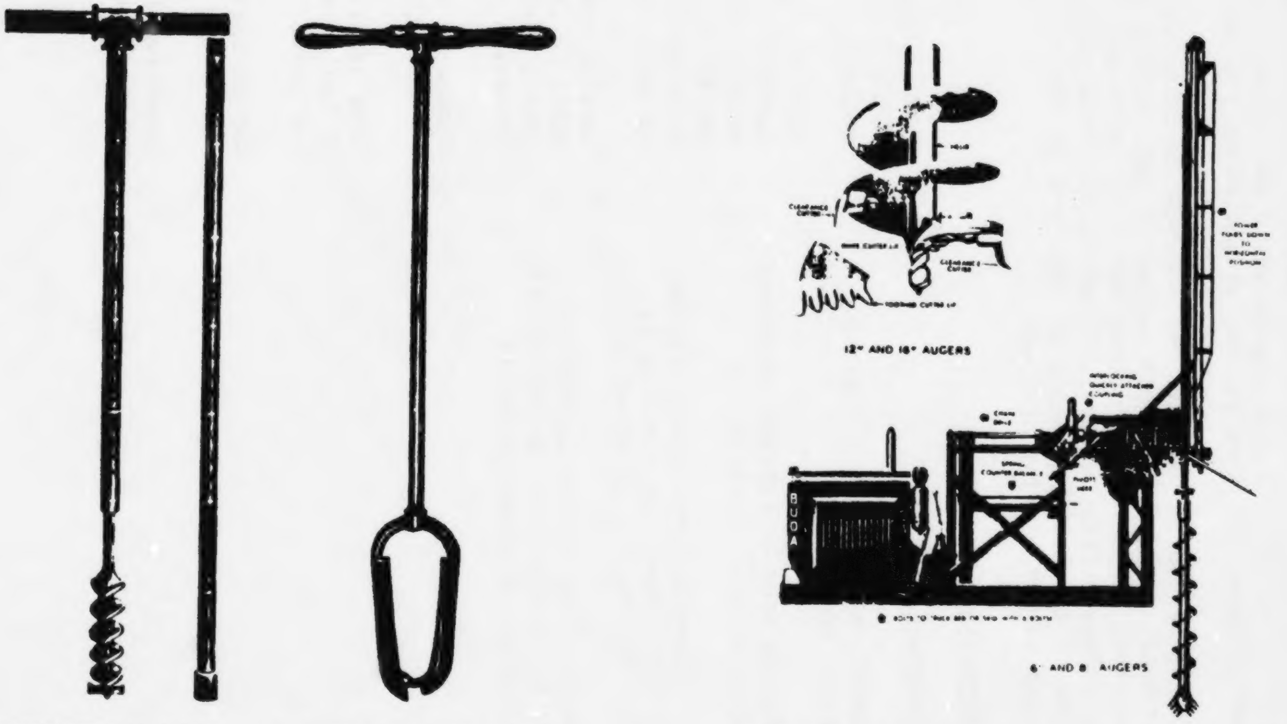
1. Inexpensive.
2. Easily installed by hand, to a limited depth.
3. Water samples can be collected at closely-spaced intervals during drilling.
4. Can expect a good seal between casing and formation. Little or no vertical leakage.

Disadvantages

1. Difficult to develop and sample if water table is below 4½ to 6 meters (15 to 20 feet).
2. Depth limitations. Applicable to shallow work primarily less than 9 meters (30 feet).
3. No formation samples, only information on subsurface material penetration rate.
4. Only certain types of pumping equipment can be used.
5. Drive point screen may become clogged with clay, if driven through a clay unit.
6. Can be used only in unconsolidated sediments.

3.5.1.2 Augers. In auger boring, the hole is advanced by rotating and pressing a soil auger into the soil and withdrawing and emptying the auger when it is full. Since water tends to prevent accumulation of soil in the auger, the borehole is kept dry as much as possible. Hand augering can be easy or difficult depending upon whether clay, sand, or gravel, respectively, is being removed. Small-diameter helical or posthole augers can be used to advance 5- to 30-cm (12-inch) diameter holes by hand to depths of 6 to 9 meters (20 to 30 feet) (Figure 46). If a tripod and pulley are set up to aid in pulling the auger from the hole, depths of 24 meters (80 feet) can be reached. If the hole can be kept open below the water table (usually only in cohesive material), screen and casing can be set, backfilled, and developed.

The process becomes much simpler and less time consuming if power augers are used. Here, flights of spiral, hollow-stem augers are forced into the ground while being rotated; the spiral action of the augers conducts cuttings to the surface (Figure 46). On completion


12" AND 16" AUGERS
BUDA
6" AND 8" AUGERS

SMALL HELICAL AUGER

POSTHOLE OR IWAN AUGER

BUDA EARTH DRILL WITH CONTINUOUS HELICAL AUGERS

FIGURE 46. AUGER EQUIPMENT

(After Hvorslev, 1965)[1]

of drilling, a small-diameter casing and well point are pushed to the desired depth. When bucket augers are used, a large diameter barrel (up to 122cm--48 inches) fitted with cutting blades is rotated into the ground until it is full. The earth-laden bucket is then brought to the surface, pulled to one side, and dumped. This process is repeated to completion depth. (Bucket augers would not normally be used in landfill investigations, and they are not evaluated in this manual.)

Power augers can be used very effectively in cohesive soils. On the other hand, they are not well suited for use in very hard or cemented soils; they often fail to retain very soft soils and fully-saturated cohesionless soils. However, if setting a drive point is the main purpose of the hole, slumping or caving-in of the hole in cohesionless sediment is not a major drawback.

Advantages

1. Inexpensive.
2. Small, high-mobility rigs can reach most sites.
3. Can be used to quickly construct shallow well clusters.
4. If borehole prematurely reaches refusal depth, set-up time is low and rig can be moved rapidly.
5. No drilling fluids introduced into the borehole; no possibility of diluting formation water.

Disadvantages

1. Limited penetration; normally 30 to 46 meters (100-150 feet) maximum.
2. Vertical leakage through sediment left in borehole through which drive point is forced to completion depth. No method to isolate screened zones of aquifer.
3. Careful attention during drilling is required to obtain correct log of formation materials penetrated.
4. Unable to collect ground-water samples during drilling.
5. Core sampling is possible only if hollow-stemmed auger flights are used.
6. Can be used in unconsolidated sediments.
7. Borehole will collapse in cohesionless sediment.

3.5.1.3 **Wash Borings.** A wash boring is advanced partly by a chopping and twisting action of a chisel-shaped bit and partly by the jetting action of a stream of water pumped through the drill rod and out the bit (Figure 47). As the bit penetrates the formations, the washing action of the bit causes the casing to sink. Cuttings are carried to the surface by the water circulating in the annular space between the drill rod and casing. The drill string is lifted and dropped, while the bit rotates, achieving a cutting action and producing a round hole. These operations, as well as the pumping, may be performed entirely by hand; but a small motor-driven winch and pump are generally used. A closed system is implemented to recirculate the drilling water. Water is pumped from a pit into the drill string and out the bit. After circulating from the bottom to the top of the borehole, the water is conducted back to the pit where the cuttings settle out.

The drill rod is generally 2.5- to 5-cm (1- to 2-inch) black-iron pipe. Casing is required to keep the hole open in soft clays or sand and gravel but is often unnecessary in stiff clays or similar cohesive sediments. If the borehole stays open by itself, casing and screen are simply lowered and backfilled to construct a well. If casing is required to drill, slip screens are set by the casing pull-back method.

Advantages

1. Inexpensive. Light equipment. Drilling contractor not required.
2. Excellent for shallow bore holes in unconsolidated sediments.
3. Can obtain vertically-spaced ground-water sample if drive point is forced ahead of borehole and pumped.
4. Drilling equipment can reach almost any site.
5. Core samples can be collected.

Disadvantages

1. Slow, especially at depth.
2. Maximum depth of 30 to 46 meters (100-150 feet).
3. Cannot penetrate boulders or wash up coarse gravel.
4. Can be used only in unconsolidated sediment.
5. Wash water can dilute formation which must be taken into account in vertical sampling.
6. Interpretation of geology from wash samples requires skill.
7. Can set only short sections of screen without difficulty.

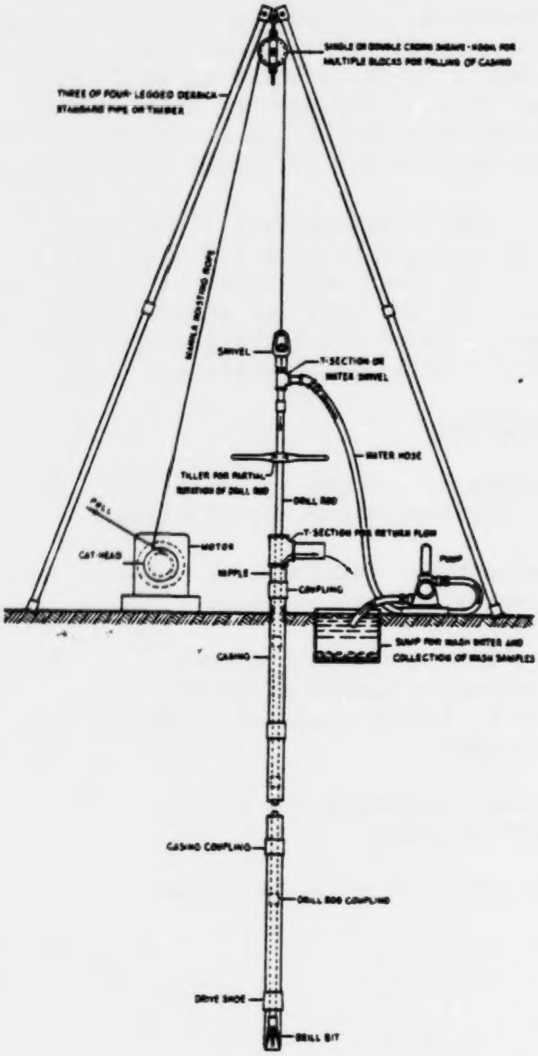

SINGLE OR DOUBLE CROWN SHEAVE - HOOK FOR MULTIPLE BLOCKS FOR PULLING OF CASING
THREE OR FOUR-LEGGED DERRICK STANDARD PIPE OR TIMBER
MANILA HOISTING ROPE
SWIVEL
T-SECTION OR WATER SWIVEL
WATER HOSE
TILLER FOR PARTIAL ROTATION OF DRILL ROD
DRILL ROD
PULL
T-SECTION FOR RETURN FLOW
MOTOR
CAT-HEAD
PUMP
NIPPLE
COUPLING
SUMP FOR WASH WATER AND COLLECTION OF WASH SAMPLES
CASING
CASING COUPLING
DRILL ROD COUPLING
DRIVE SHOE
DRILL BIT

FIGURE 47. SIMPLIFIED WASH BORING RIG

(After Hvorslev, 1965)[1]

3.5.1.4 **Jet Percussion.** The drill tools and the drilling action of the jet-percussion method are the same as those described for the wash-boring method. One essential difference, however, is that during drilling the casing is driven with a drive weight and not allowed to advance of its own weight. Normally, this method is used to place 5-cm (2-inch) diameter casing in shallow, unconsolidated sand formations but has been used to install 7.6- to 10-cm (3- to 4-inch) diameter casings to 61 meters (200 feet). Screens should be set by the casing pull-back method. Most jet-percussion rigs (Figure 48) are moderately-sized units. Drilling contractors used to working in unconsolidated sediments will probably be the best source for a rig.

Advantages

1. Relatively inexpensive.
2. Simple equipment and operation.
3. Good seal between casing and formation prevents vertical leakage of formation water.
4. Can obtain a reliable formation water sample at completed depth.

Disadvantages

1. Slow.
2. Use of water during drilling can dilute formation water.
3. No formation water samples can be taken during drilling.
4. Poor soil samples due to the fact that fines are washed out of sample.
5. Large number of wells is required at one location to obtain closely-spaced samples throughout the contaminated thickness of the aquifer.
6. Can only be used on unconsolidated sediments or weathered rock.

3.5.1.5 **Cable-Tool (Percussion).** In cable-tool or percussion drilling, the hole is deepened by regularly lifting and dropping a heavy string of drilling tools in the borehole (Figure 49). The drill bit breaks or crushes hard rock into small fragments and in soft, unconsolidated sediments loosens the material. The up and down action of the drill string mixes the crushed or loosened particles with water to form a slurry. If no water is present in the formation being penetrated, water is added to the borehole. Cuttings are allowed to accumulate until they start to lessen the impact of the bit and then are removed with a bailer or sand pump.

A cable-tool drill string consists of three units:

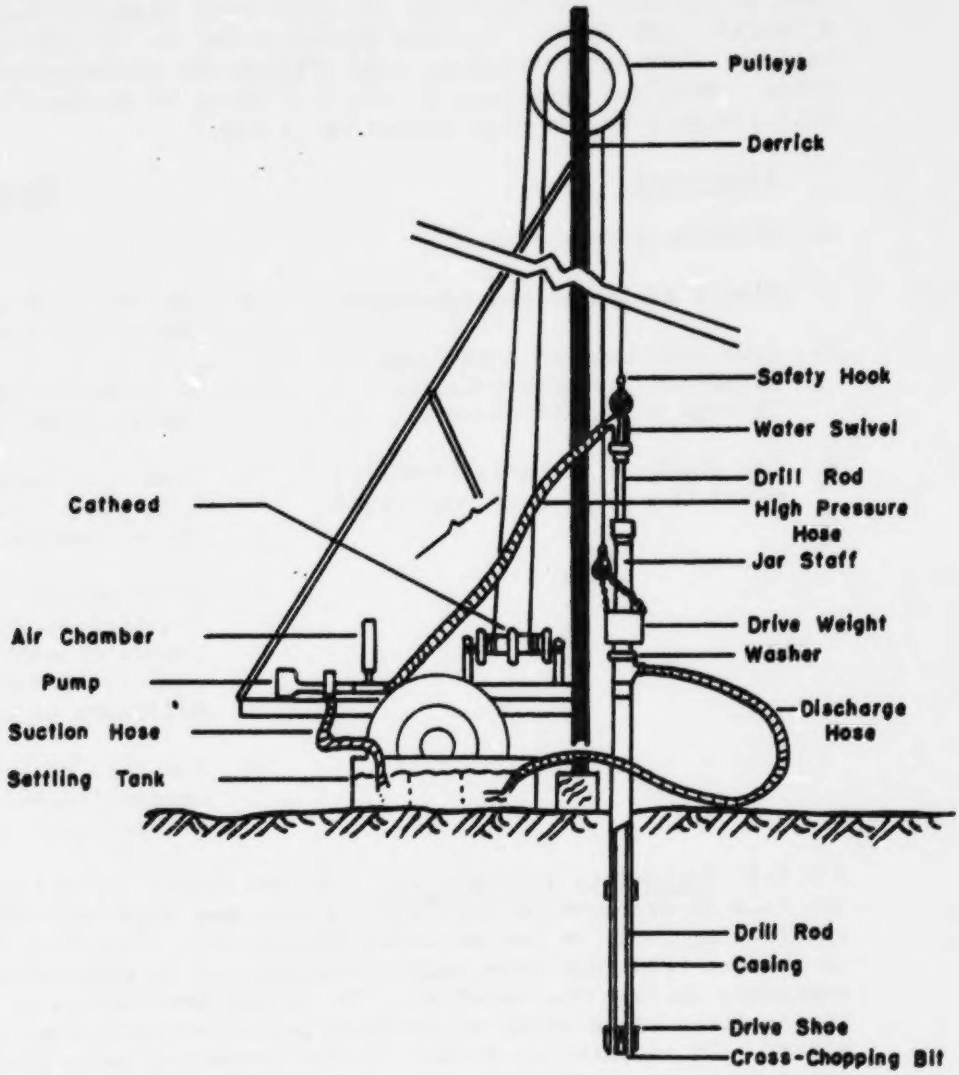

Pulleys
Derrick
Safety Hook
Water Swivel
Drill Rod
High Pressure Hose
Jar Staff
Cathead
Drive Weight
Washer
Air Chamber
Pump
Suction Hose
Discharge Hose
Settling Tank
Drill Rod
Casing
Drive Shoe
Cross-Chopping Bit

FIGURE 48. SIMPLIFIED JET - PERCUSSION

(After Matlock, 1970)

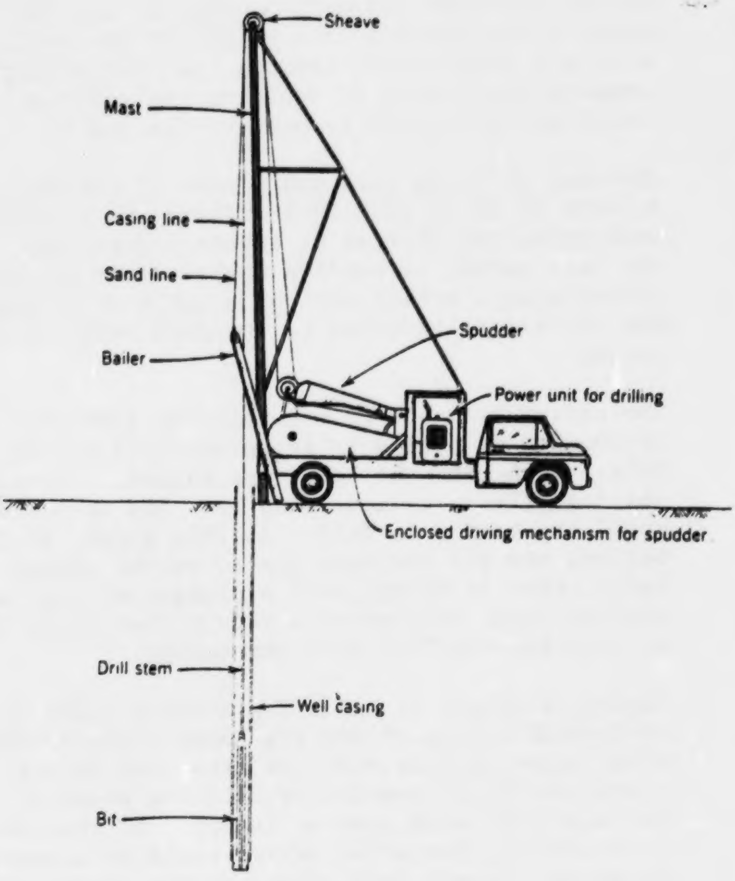

Sheave
Mast
Casing line
Sand line
Spudder
Bailer
Power unit for drilling
Enclosed driving mechanism for spudder
Drill stem
Well casing
Bit

FIGURE 49. SIMPLIFIED CABLE TOOL PERCUSSION RIG

(After Davis & De Wiest, 1966)

- the drill bit,
- drill stem,
- rope socket.

The bit provides the cutting edge of the drill string, the action of which is increased by the weight of the drill stem. This weight also acts as a stabilizer, keeping the hole straight. The rope socket connects the string of tools to the cable and allows the tools to rotate slightly with respect to the cable.

Optional drilling jars are a pair of sliding, linked bars which provide a slack of 15 to 23cm (6-9 inches) in the drill string. These are used primarily in clay or caving formations. If the tools become stuck, the jars permit successive upward blows in the attempt to free them rather than a steady pull on a cable which might part. The shaking and vibrations produced by the jars helps in freeing a stuck drill string.

The bailer consists of a section of pipe with a check valve at the bottom and is filled by an up-and-down motion in the bottom of the hole. Each time the bailer is dipped, the valve opens, allowing the cuttings slurry to move into it. The up-and-down motion is continued until the bailer is full. At this point, it is brought to the surface and the contents dumped on the ground. The sand pump is a bailer that is fitted with a plunger so that an upward pull on the plunger tends to produce a vacuum that opens the valve and sucks sand or slurried cuttings into the tubing.

Casing is driven by attaching a drive clamp to the drill stem; the reciprocal action of the rig hammers the casing into the ground as the clamp makes contact with the drivehead on top of the casing. The operation can be speeded by drilling ahead of the casing but only if the hole will stay open by itself. If when drilling an open hole there is a cave in, the drill string could be trapped. Cautious drillers, therefore, rarely drill ahead of the casing unless they are going through rock. Normal procedure in unconsolidated sediments is to drive the casing into the formation and then to clean out inside the casing with the drill tools. This is slower but safer than drilling ahead of the hole.

Advantages	Disadvantages
1. Simple equipment and operation.	1. Slow.
2. Good seal between casing and formation if flush joint casing is used.	2. Use of water during drilling can dilute formation water.

3. Good disturbed soil samples. Known depth from which cuttings are bailed.

4. Core samples can be collected.

5. If casing can be bailed dry without sand heaves, a formation-water sample can be collected.

6. Can be used in unconsolidated sediments and consolidated rocks.

7. Only small amounts of water are required for drilling.

8. Once water is encountered, changes in static or potentiometric levels are readily observable.

3. Potential difficulty in pulling casing in order to set screen.

4. No formation water samples can be taken during drilling unless open-ended casing is pumped, or a screen set.

5. Heavy steel drive pipe is used and could be subject to corrosion under adverse contaminant characteristics.

6. Cannot run a complete suite of geophysical well logs because of casing.

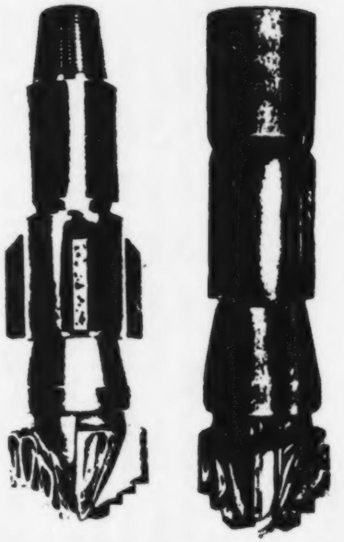
3.5.1.6 Hydraulic Rotary. In drilling a hole by the hydraulic rotary method, a rotating bit breaks up the formation and the cuttings are brought to the surface by a recirculating drilling fluid (Figure 50). Drilling mud is pumped from a settling basin, through a water swivel, down the hollow interior of the drill rod, and through the bit. The fluid then flows upward in the annulus, carrying the drill cuttings to the surface. Here, it is discharged into the mud pit, and the cuttings settle out. At the other end of the pit, the fluid is pumped out to circulate down the drill rod again.

The drill string consists of the bit, a stabilizer, and the drill pipe. Two basic types of bits are used:

- roller bits in rock,
- drag bits in unconsolidated materials.

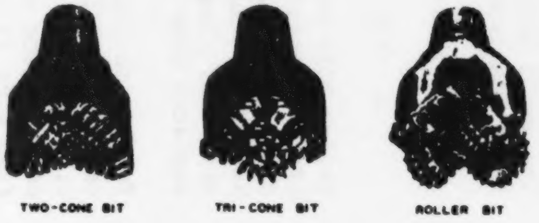
Roller bits have conical rollers with hardened steel teeth of various lengths, spacing and shape dependent upon the type of material to be drilled. Some rollers have inset carbide buttons for drilling in hard rock. As the rollers rotate, they crush and chip the formation material. Drag bits have fixed blades, and the cutting edge is surfaced with carbide or some other abrasion-resistant material.

The bit is attached to a heavy, weighted section of the drill string called a drill collar or stabilizer. This weight, just above the bit, tends to keep the borehole straight and vertical. The drill rod connects the stabilizer to the kelly. The outside diameter ranges from 6 to 11.4cm (2 3/8 - 4½ inches). The kelly is a fluted or square



DRAG-TYPE BITS WITH REPLACEABLE BLADES.

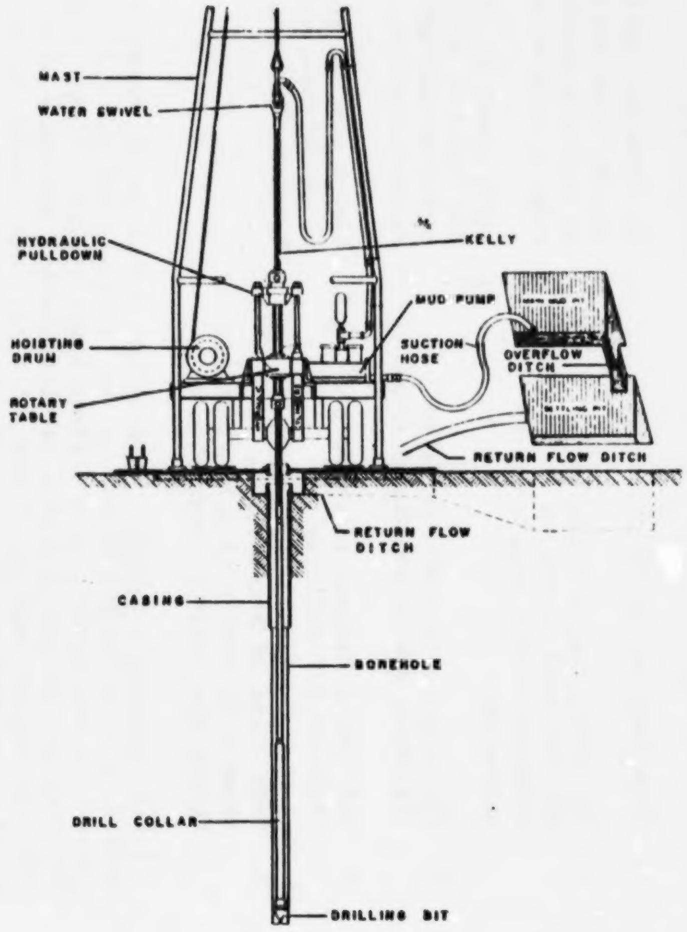
(After Johnson, 1972)



TWO-CONE BIT
TRI-CONE BIT
ROLLER BIT

ROCK BITS

(After Hvorslev, 1965)



MAST
WATER SWIVEL
HYDRAULIC PULLDOWN
HOISTING DRUM
ROTARY TABLE
KELLY
MUD PUMP
SUCTION HOSE
OVERFLOW DITCH
RETURN FLOW DITCH
RETURN FLOW DITCH
CASING
BOREHOLE
DRILL COLLAR
DRILLING BIT

(After Hvorslev, 1965)

FIGURE 50. HYDRAULIC ROTARY DRILLING EQUIPMENT

bar which passes through a rotary table and imparts a rotary motion to the drill string. When the length of the kelly has been drilled, a new section of rod is added and drilling resumed.

Advantages

1. Fast.
2. Dilution of formation water is limited by formation of a filter cake on borehole walls.
3. Formation water sample can be obtained with a special technique.
4. Good disturbed soil samples from known depths if travel time of borehole cuttings is taken into account, although sorting may occur.
5. Flexibility in final well construction.
6. Can run a complete suite of geophysical well logs.
7. Core samples can be collected.
8. Can be used in unconsolidated sediments and consolidated rocks.

Disadvantages

1. Expensive.
2. Requires complex equipment and operation.
3. There is a potential for vertical movement in formation stabilizer material placed between casing and borehole wall after completion.

3.5.1.7 Air Rotary. With this drilling method, a rotating down-hole hammer may be used to break up formation material by percussion, although conventional rotary bits are also used. However, rather than carrying cuttings to the surface, high velocity compressed air is used. Down-hole hammers are essentially the pneumatic hammer type, similar in operation to those used by road repair crews to break up pavement. Since the penetration rates of the air rotary drilling method are much greater, this type of drilling is favored over the cable-tool or hydraulic-rotary drilling. Drilling rates of 30 to 60cm (1-2 feet) per minute are not unusual. One disadvantage, however, is that down-hole hammers larger than 20cm (8 inches) are not readily available, limiting the size of the borehole that can be drilled. Much of their speed advantage is lost when conventional roller cone bits are used. Most rigs are equipped with a small mud pump permitting a conventional rotary hole to be drilled through unconsolidated overburden to competent rock. When this hole is finished, casing is set into rock to prevent caving, and drilling continues using the rotary method.

Consequently, less water is required, thereby reducing a logistics problem that can become difficult, especially in arid regions.

A minimum upward air velocity of 915 meters (3,000 feet) per minute is required to lift cuttings to the surface. When drilling a 10-cm (4-inch) diameter hole with 6-cm (2 3/8-inch) rod, at least 150 cubic feet per minute (CFM) of air are required to lift cuttings. If a prolific aquifer is penetrated, the compressed air may be unable to lift the volume of water entering the hole to the surface. At 915 meters (3,000 feet)perminute air velocity, this threshold is met at about 3.15 liters/sec (50 gpm) in a 10-cm (4-inch) hole and at about 9.5 liters/sec (150 gpm) in a 15-cm (6-inch) hole. When this happens, a larger air compressor is required or drilling must be changed to the hydraulic rotary method. Air rotary rigs are available with compressors capable of supplying 31 cubic meters/minute (1,100 CFM) at a pressure of 17.58 kg/cm^2 (250 psi).

Advantages	Disadvantages
(See section on Hydraulic Rotary Drilling--Section 3.5.1.6)	

3.5.2 Well Casing and Screen Materials

Landfill leachate can be characterized as a strong electrolyte which may be corrosive. Specific characteristics of the leachate will depend upon the type of material accepted by the operators. Therefore, some thought must be given to the materials used in monitoring well construction to prolong the installation's operating life. Review of comparison tables of various pipe materials to chemical attack indicates that PVC pipe is resistant to most chemicals, with the exception of ketones, esters, and aromatics (among the more common chemicals) when compared to steel well casing. PVC casing is a nonconductor and will not be involved in electrochemical reactions as will a steel casing and brass or iron well screen.

From a leachate sampling standpoint, PVC is very attractive. Because of its chemical inertness, it will contribute little in the way of chemical constituents to a leachate sample except in the parts-per-billon range. Steel pipe can be expected to contribute at least iron and probably other ions to a sample. Of course, this sample contamination can probably be avoided by proper flushing of the well before collection.

A major drawback to PVC casing is its lack of strength. PVC casing projecting above ground surface can be easily broken by landfill equipment or vandals. Therefore, special well-protection measures (described in the Well Security section--3.5.3) must be taken. In spite of this, PVC casing and screen appear to be the best materials to use in constructing landfill monitoring wells.

Actual well construction, however, will be dictated by a variety of constraints, such as:

- contaminants being sampled for,
- drilling methods,
- aquifer type and formation materials,
- cost of well construction materials,
- ease of installation.

Proper construction materials can be best evaluated for each situation by a person familiar with landfill investigations, but a person not familiar with them can fall back on a drilling method that will allow PVC casing and screen to be installed and can be confident that the well will last and not bias the samples.

3.5.3 Well Security

Once a well has been completed, some measures must be taken to protect the installation from normal landfill operations (especially the use of heavy equipment) and from vandals. In areas actively landfilled, provisions should be made for extending the well casing and its protection above the active level of the fill. An installation capable of protecting the monitoring well and also capable of being added on to the depth of the fill increases is illustrated in Figure 51.

Construction of this protective installation is straightforward and inexpensive with a reasonable likelihood of remaining undamaged by landfill equipment or vandals. To accomplish this, a 3-meter (10-foot) length of steel casing several centimeters (inches) larger in diameter than the monitoring well is emplaced. This casing is grouted and placed with a cement collar at least 1.2 to 1.5 meters (4-5 feet) deep to hold it firmly in position. Although not sufficiently rugged to withstand a run-in with a compactor or bulldozer, it will withstand attempted vandalism. The casing should be threaded so that a screw cap can be used to close the well. Two heavy-duty hardened steel hasps welded on opposite sides of the cap and casing will allow the well to be locked.

Unless the well is highly visible, the probabilities of it being struck by equipment during normal landfill operations are great. To avoid this, a simple tripod constructed of timbers (railroad ties or equivalent) should be constructed over the well and crowned with a brightly colored object such as a flag or painted tire. The well could also be protected by brightly painted drums filled with soil.

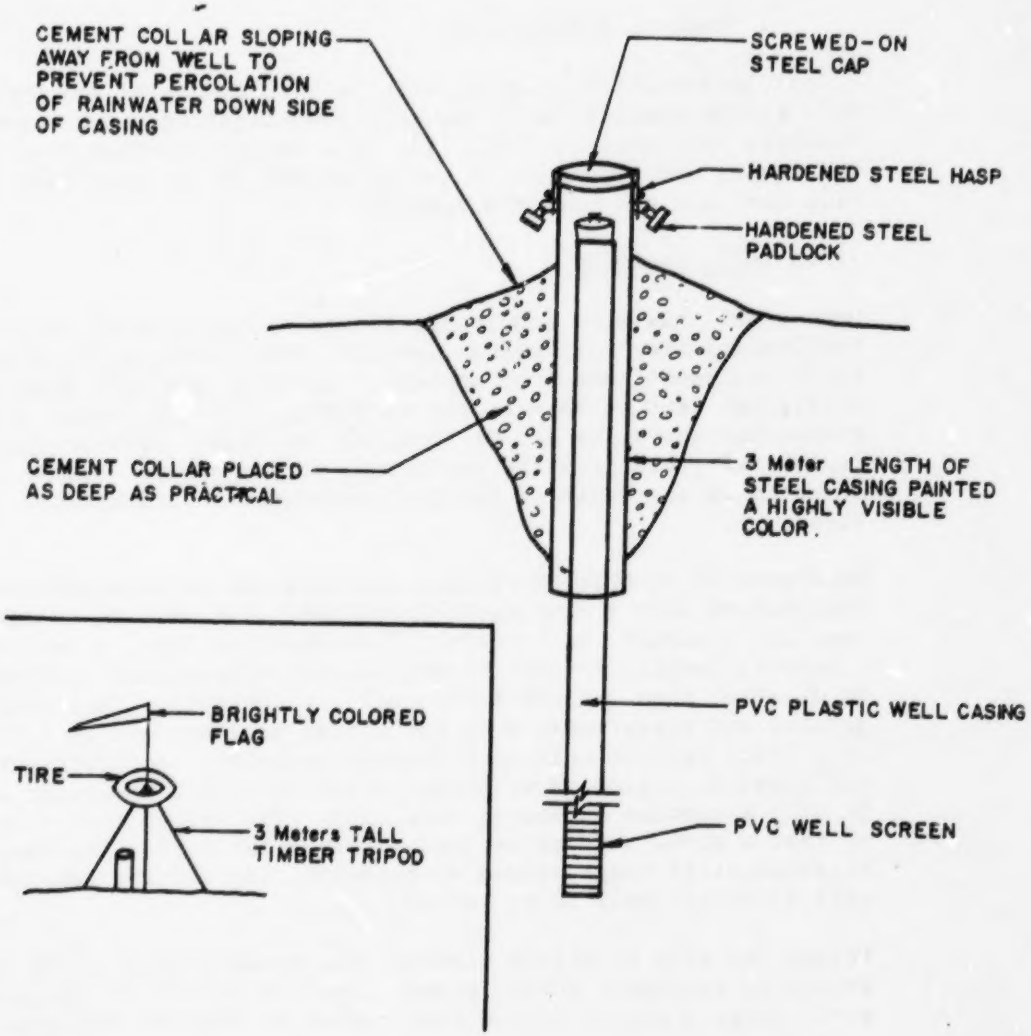

CEMENT COLLAR SLOPING AWAY FROM WELL TO PREVENT PERCOLATION OF RAINWATER DOWN SIDE OF CASING
SCREWED-ON STEEL CAP
HARDENED STEEL HASP
HARDENED STEEL PADLOCK
CEMENT COLLAR PLACED AS DEEP AS PRACTICAL
3 Meter LENGTH OF STEEL CASING PAINTED A HIGHLY VISIBLE COLOR.
BRIGHTLY COLORED FLAG
TIRE
3 Meters TALL TIMBER TRIPOD
PVC PLASTIC WELL CASING
PVC WELL SCREEN

FIGURE 51. GENERALIZED METHOD FOR PROTECTING A WELL OR PIEZOMETER

When landfilling approaches the top of the installation, the tripod is temporarily knocked down, additional casing added to the monitoring well and protective shell, and placement of trash and cover is continued around the well. If this procedure is followed, only a slight interruption in the normal course of landfill operation will be required to protect the monitoring well for future sampling.

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4.0 INDICATORS OF LEACHATE

4.1 INTRODUCTION

Leachate represents an extremely complex system containing soluble, insoluble, organic, inorganic, ionic, nonionic, and bacteriological constituents in an aqueous medium. Figure 52 schematically depicts an extensive characterization of leachate by means of physical, inorganic, bacteriological, and organic parameters. Chapter 5 presents additional information on leachate characteristics.

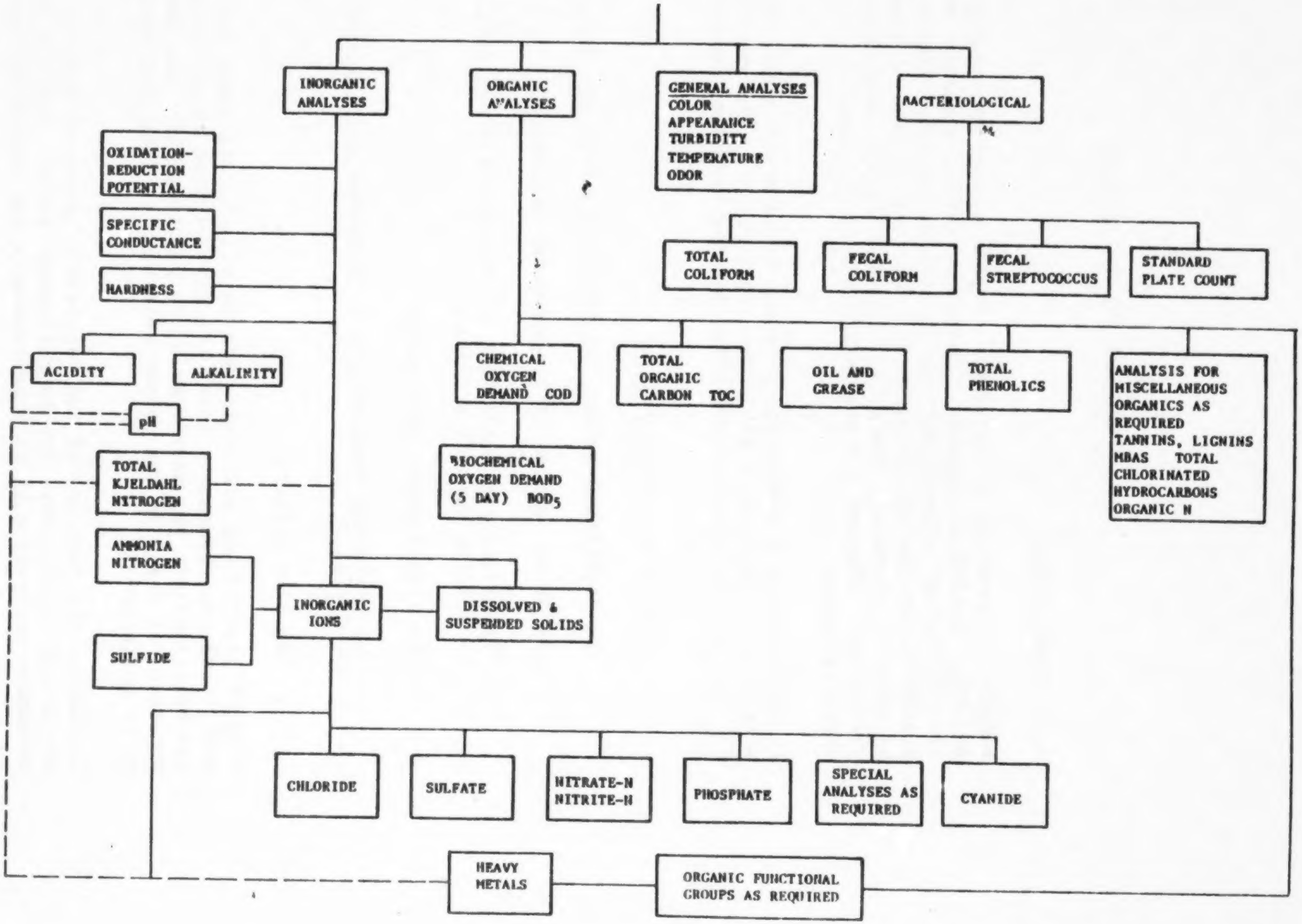
In establishing a monitoring program, consideration should be made of all the factors affecting the quality of the pure leachate and leachate-enriched ground water and the resultant environmental impact, including:

- purpose for monitoring;
- the background quality of ground water at the site;
- the other sources of ground-water pollution;
- the hydrologic conditions of the site and the resultant monitoring network utilized;
- the climatologic influences;
- the costs and availability of manpower and laboratory facilities;
- the site history.

A major item in any monitoring system will be the costs for the analytical measurements. There are several ways these costs may be minimized, yet meet regulatory requirements. The most important of these is the proper selection of indicator parameters to be monitored.

It will be the function of the regulatory agency to specify monitoring requirements for land disposal sites. Some of the necessary analyses will be time-consuming and relatively expensive.

FIGURE 52. SCHEMATIC DIAGRAM OF LEACHATE CHARACTERIZATION


INORGANIC ANALYSES
ORGANIC ANALYSES
GENERAL ANALYSES
COLOR
APPEARANCE
TURBIDITY
TEMPERATURE
ODOR
BACTERIOLOGICAL
OXIDATION-REDUCTION POTENTIAL
SPECIFIC CONDUCTANCE
HARDNESS
TOTAL COLIFORM
FECAL COLIFORM
FECAL STREPTOCOCCUS
STANDARD PLATE COUNT
ACIDITY
ALKALINITY
pH
CHEMICAL OXYGEN DEMAND COD
TOTAL ORGANIC CARBON TOC
OIL AND GREASE
TOTAL PHENOLICS
ANALYSIS FOR MISCELLANEOUS ORGANICS AS REQUIRED TANNINS, LIGNINS MBAS TOTAL CHLORINATED HYDROCARBONS ORGANIC N
BIOCHEMICAL OXYGEN DEMAND (5 DAY) BOD5
TOTAL KJELDAHL NITROGEN
AMMONIA NITROGEN
SULFIDE
INORGANIC IONS
DISSOLVED & SUSPENDED SOLIDS
CHLORIDE
SULFATE
NITRATE-N NITRITE-N
PHOSPHATE
SPECIAL ANALYSES AS REQUIRED
CYANIDE
HEAVY METALS
ORGANIC FUNCTIONAL GROUPS AS REQUIRED

The regulatory agencies should maintain flexibility to consider approval for substitution of a less expensive analytical indicator if the parameter requiring the more expensive analysis can be accurately inferred from the simpler, less expensive analytical indicator. For example, the substitution of the COD (Chemical Oxygen Demand) analysis for a portion of the BOD_5 (5-day Biochemical Oxygen Demand) analysis should be allowed in those states where the latter is required if it can be shown that a satisfactory correlation exists between the two parameters. Monitoring land disposal sites can also be viewed in the sense of quality control. A regulatory agency could allow (at least for frequent monitoring) that the selection of indicators be subject only to the requirement of quality-control comparisons between ground-water quality and permit specifications. It is here that a strong effort should be made to utilize inexpensive indicator analyses which can provide quick, accurate, and correct information of those indicators requiring more expensive analytical techniques. Promptness of analysis is quite important since having the results for early action will greatly simplify control requirements and sample degradability effects will be minimized. As an example, conductivity can sometimes be used as an indication of total dissolved solids. This is a simple measurement and one which gives immediate results for quality-control comparisons. However, for quantitative extrapolations it is absolutely necessary to obtain a correlation between the two indicator analyses for the particular land disposal site being monitored.

This chapter will provide guidelines for selecting indicators as well as scheduling and data management and interpretation resulting in a representative, valid, and cost/effective monitoring program. The emphasis of this chapter is on monitoring of ground-water sampling devices, strategically located in reference to ground-water flow directions at regular intervals to determine chemical constituents in the ground water at that point and time. In addition, this chapter assumes that the landfill being monitored disposes only normal municipal solid waste. Where special wastes are involved, such as hazardous chemical and liquid wastes, the indicator selection and sample scheduling would be modified accordingly to be more waste specific.

4.2 BACKGROUND QUALITY OF THE GROUND WATER

The background-water quality at a land disposal site must be considered in selecting the indicators for a monitoring program. A ground water with a high background concentration of a particular parameter would certainly lessen the value of that parameter as a leachate indicator as it will require higher concentrations to differentiate from background.

No rule of thumb can be offered as to what background concentration levels render a parameter useless as a leachate indicator. Such a determination will be relative to the specific parameter. Its limits of detection, the concentration difference observed between the background and downgradient monitoring wells and the standards of the regulatory agency must be considered. Phenols and total dissolved solids (TDS) are presented as two extreme examples on this matter. The drinking water standard for phenols is 1 part per billion (ppb) which is also its limit of detection. Unless the downgradient monitoring well contains a very high concentration of phenols, even the slightest background concentration of phenols would interfere with data interpretation. However, the drinking water standards for TDS is 500 parts per million (ppm) orders of magnitude higher than its limits of detection. Therefore, high background concentrations of TDS can be tolerated without interfering with data interpretation.

In a given landfill situation, it is necessary to obtain adequate background data in order to draw reliable conclusions regarding possible leachate contamination. Reliable data on background quality of ground water will also be of critical importance relative to regulatory and legal considerations. Therefore, consideration must be given to both the ground-water quality which occurs naturally, as well as other possible sources of contamination which may affect the background quality.

4.2.1 Chemical Quality of Natural Ground Water

All ground water contains chemical constituents in solution. The kinds and amounts of constituents depend upon the geologic environment, movement, and source of the ground water. Typically, concentrations of dissolved constituents in ground water exceed those in surface waters. This is particularly true in arid regions where recharge rates are low.

Dissolved constituents are primarily derived from minerals in contact with ground water and percolating water going to ground-water recharge. Common chemical constituents of ground water include:

Cations

Calcium
Magnesium
Sodium
Potassium

Anions

Carbonate
Bicarbonate
Sulfate
Chloride
Nitrate

Undissociated

Silica

Table 4 lists relative abundances of these and other chemical constituents in natural ground water. Minor and trace constituents

TABLE 4

RELATIVE ABUNDANCE OF DISSOLVED SOLIDS IN POTABLE WATER

Major Constituents (1.0 to 1000 ppm)

Sodium
Calcium
Magnesium
Bicarbonate
Sulfate
Chloride
Silica

Secondary Constituents (0.01 to 10.0 ppm)

Iron
Strontium
Potassium
Carbonate
Nitrate
Fluoride
Boron

Minor Constituents (0.001 to 0.1 ppm)

Antimony
Aluminum
Arsenic
Barium
Bromide
Cadmium
Chromium
Cobalt
Copper
Germanium
Iodide
Lead
Lithium
Manganese
Molybdenum
Nickel
Phosphate
Rubidium
Selenium
Titanium
Uranium
Vanadium
Zinc

Trace Constituents (generally less than 0.0001 ppm)

Beryllium
Bismuth
Cesium
Gallium
Gold
Indium
Lanthanum
Platinum
Silver
Thallium
Thorium
Tin
Tungsten
Zirconium

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*Davis, F. N., and R. J. M. De Wiest. Hydrogeology. John Wiley & Sons, New York, 1966.

are present selectively depending upon the mineralogy of the region. Analyses of ground-water samples enriched in silica, iron, calcium, and sodium are given in Table 5. These elements are frequently enriched in ground water. Brines and thermal spring waters are not included in Table 5.

For convenience, ground-water quality is classified on a simplified basis according to domestic and industrial use. Salinity (the concentration of total dissolved solids) and hardness (the combined calcium and magnesium concentrations) are classificatory criteria. The classification scheme is shown on Table 6. Water containing a high concentration of dissolved solids can build up scale in boilers, be harmful to plants when used for irrigation, and interfere with quality of products in manufacturing. Hard water also builds up scale deposits in boilers and forms scums with soap in laundering.

Within a large body of ground water, the natural chemical composition tends to be relatively consistent. Variation of ground water relative to time is minor in comparison with surface-water quality changes.

Under natural conditions, salinity of the ground water tends to increase with depth. Most of the geologic formations in the United States are underlain by brackish to highly saline waters. Density and permeability differences act to maintain a separation between these waters and the overlying fresh ground water.

4.2.2 Other Sources of Ground-Water Contamination

Ground-water composition can vary widely under natural conditions. Man's activities add another dimension to the complexity of ground-water quality. The effect on ground-water quality of point sources of contamination, such as: waste laggons, acid mine spoils, and oil well brines are relatively easy to trace. Diffuse sources of contamination, such as: regions of septic tanks, irrigation, or farm chemical usage may affect bodies of ground water, creating relatively uniform enrichment. Detection of point-source contamination within a region of general contamination will require relatively higher concentrations of contaminants in comparison to comparable uncontaminated areas.

No estimate of background concentrations of leachate indicators typical for a geohydrological setting could be used in lieu of actual measurements. The only way to ascertain the ground-water quality at a given site is to measure it. As previously mentioned, man's activities can influence ground-water quality. A description of contaminants associated with certain of these activities would be helpful as it points out those indicators which need to be

TABLE 5

ANALYSES OF GROUND WATER IN WHICH THE INDICATED ELEMENT IS A MAJOR CONSTITUENT. (CONCENTRATIONS IN MG/L UNLESS NOTED.)*

Constituent	Silica	Iron	Calcium	Sodium
Silica (SiO_2)	99	20	29	16
Aluminum (Al)	--	--	--	--
Iron (Fe)	0.04	2.3	--	0.15
Calcium (Ca)	2.4	126	636	3
Magnesium (Mg)	1.4	43	43	7.4
Sodium (Na)	100	13	17	857
Potassium (K)	2.9	2.1		2.4
Carbonate (CO_3)	24	--	--	57
Bicarbonate (HCO_3)	111	440	143	2,080
Sulfate (SO_4)	30	139	1,570	1.6
Chloride (Cl)	10	8.0	24	71
Fluoride (F)	22	0.7	--	2.0
Nitrate (NO_3)	0.5	0.2	18	0.2
Dissolved solids				
Calculated	348	594	2,410	2,060
Residue	---	571	--	--
Hardness as $CaCO_3$	12	490	1,760	38
Noncarbonate	0	131	1,650	0
Specific condustance umhos/cm @25	449	885	2,510	2,960
pH units	9.2	7.6	--	8.3
Temperature, °C	50.0	13.3	--	--

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*Hem, John D. Study and interpretation of the chemical characteristics of natural water. In U.S. Geological survey water supply paper No. 1473, 1970.

TABLE 6

CLASSIFICATION OF WATER *

Name	Concentration of total dissolved solids ppm
Based on salinity	
Fresh	0-1000
Brackish	100-10,000
Salty	10,000-100,000
Brine	over 100,000
Slightly saline	1000-3000
Moderately saline	3,000-10,000
Very saline	10,000-35,000
Briny	over 35,000
Based on hardness	Hardness as $CaCO_3$ ppm
Soft	0-60
Moderately hard	61-120
Hard	121-180
Very hard	over 180

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*Hem, John D. Study and interpretation of the chemical characteristics of natural water. In U.S. Geological survey water supply paper No. 1473, 1970.

delineated from unnaturally high background concentrations in order to trace leachate-enriched ground water.

The following sources of ground-water contamination are listed in Table 7 which summarizes their potential contributions of ground-water contaminants. Their similarities to and distinctions from leachate should be carefully noted so that interferences will be recognizable.

4.2.2.1 Highway Deicing. Over 6,567,000 tons (5,962,000 tonnes) of deicing salts were used nationwide in 1966-67.[1] The most common salt in use is sodium chloride. Calcium chloride use amounts to only 4 percent of that of sodium chloride.

Open storage of salt or salt/sand mixutres may result in leaching of salt with rainwater. The leachate after reaching ground water will form a plume of salt-enriched ground water which could contaminate wells in the vicinity. However, spreading of salt on the road results in a more diffuse salt-enrichment of ground water. Wells located near major highways have been affected by deicing salt.

4.2.2.2 Leaky Sewers. Sewer pipes which have been in service over a period of years are likely to be leaky. Sewage gases form acids which dissolve concrete and mortar, the usual substance of older sewer pipes. When the pipes are located in the unsaturated zone, raw sewage may leak and percolate to the ground water.

Sewage contains some inorganic salts, sulfur, nitrogen, trace metals, and suspended and dissolved organic compounds. Sulfur and nitrogen are generally present as sulfide and ammonia. After entering the zone of aeration, these ions are oxidized to sulfate and nitrate. Thus, sulfate and nitrate are associated with leaky sewers in Table 7. The organic matter exerts a large BOD and COD. Enteric organisms, bacteria and viruses, are present in large numbers creating a potential for biological contamination. It has been estimated that approximately 1.9 billion liters (500 million gallons of sewage is lost annually in the U. S. through leakage.[2])

TABLE 7

CONTRIBUTION OF LANDFILL LEACHATE INDICATORS TO
GROUND WATER BY OTHER SOURCES

Indicator	Highway deicing	Leaky sewers	Septic tanks	Mining	Irri-gation	Land dis-posal sludge	Petro-leum expl & dev.	Feed-lots
Phosphate		M	M	P		P		P
Calcium	M				M	L	M	
Magnesium					M	L		
Sodium	H	L	L		M	L	H	
Potassium					M			
Ammonium		M	M					
Chloride	H	L	L		H	L	H	L
Sulfate		M	M	H	M		M	
Nitrate		H	H		L	H	M	H
Bicarbonate		M	M					
Iron		H	M	H		H		
Manganese				H		H		
Boron		L	L					
Selenium								
Zinc				M		M		L
Copper				M		L		
Lead				M		L		
Other h. m.				P		P		P
MBAS		M	L					
Phenols		P				P		
PCB						P		
Org N		M	M			M		L
PAH-HC						P		
TOC		H	H			L		M
BOD		H	M					M
Coliform		P	P			P		P
Virus		P	P			P		P

H=High M=Moderate L=Low P=Potential

4.2.2.3 Septic Tanks. Contaminants carried to ground water in percolating septic tank effluent are similar to those from leaky sewers. The major difference is that septic tanks have provided an opportunity for some anaerobic decomposition. Thus, MBAS and BOD levels are reduced from those of raw sewage. Again, percolation of effluent through the zone of aeration converts ammonium and sulfide to nitrate and sulfate. A reduction of dissolved oxygen in the percolate or ground water by a high BOD flow can cause dissolution of iron, hence the iron rating in Table 7.

Unless the septic tank is within a few feet of the water table, bacterial contamination should not be a problem. However, a septic tank located in coarse-textured soil overlying a fractured rock aquifer might cause considerable bacterial and viral contamination.

Septic tanks in a density of approximately two per hectare (one per acre) or less are not likely to significantly influence regional ground-water quality. As density increases, these point sources of contamination blend together and result in a general degradation of regional ground water.

4.2.2.4 Mining. A variety of contaminants are generated in leach-mining and ore beneficiation which are specific to the mineral type and mine location. Locations generating these contaminants would be obvious, thus it is unnecessary to deal with them here. A more general contamination problem is generated by wastes from strip and shaft mining, particularly as these methods pertain to coal. In strip mining, overburden must be removed to expose coal or ore seams. The coal is separated from waste rock and is washed. The wastes produced in these processes are termed spoils and gob piles.

Frequently, the waste rock and mineral contains pyrite (FeS_2), an iron sulfide mineral. When exposed to air, pyrite oxidizes with the help of iron-oxidizing bacteria. The oxidation produces sulfuric acid which keeps iron in solution and frequently dissolves other heavy metals from waste minerals. Drainage water from strip mine spoils or from mine shafts may have a pH of less than 2. This acid mine drainage kills fish in surface waters and produces an unsightly red-yellow scum.

The acid character of the spoils and gob piles prevents growth of plant life on their surface. The resulting erosion continuously exposes new pyrite to oxidation. Acid mine drainage introduces millions of tons of sulfuric acid into the environment each year. Large regions of Pennsylvania, West Virginia, Ohio, Indiana, Illinois have been contaminated from coal mining wastes. Abandoned metal ore mines have produced a similar problem in Colorado.

Acid water can contaminate ground water with a variety of heavy metals which it dissolves. Reduction of the ground-water pH also occurs creating a more corrosive medium. Iron and/or manganese accompanying acidic water add a metallic taste, and cause staining of plumbing fixtures and laundry.

4.2.2.5 Irrigation. Ground or surface water used for irrigation becomes more mineralized as it percolates through soil and dissolves mineral and fertilizer constituents. Irrigation water going to recharge carries an enrichment of some or all of the following ions: calcium, magnesium, sodium, potassium, chloride, sulfate, and nitrate.

Continued irrigation increasingly mineralizes ground water. This enrichment may become limiting to further ground water use. In California, there are closed basins where ground water has been recycled by irrigation and has become so mineralized it is approaching the limit of usefulness.

4.2.2.6 Land Disposal of Sludge. The literature on sludge chemistry has reported all of the indicators listed in Table 7 as being present in one sample or another. Concentrations range from parts per billion to percentages.

Sludge is applied to the land surface. Therefore, its influence on ground-water quality is determined by the transport of its constituents through the top soil and underlying unsaturated zone. The contribution of landfill leachate indicators to ground water is calculated on the basis of sludge leachate having undergone reactions in the top soil and unsaturated zone.

Ammonium in sludge will nitrify, and the leaching portion will migrate as nitrate. Most of the heavy metals and phosphate will probably be retained in the soil and be in extremely low concentrations in percolate.

Bacteria have been studied after sludge application to land. Fecal coliforms exhibit a die-off rate which reduces their number to a negligible population in a matter of 2-3 weeks. Bacterial movement through soil is usually no more than a few centimeters. Viruses have been shown to be more in soil but are not likely to be a serious contaminant if digested sludge is used.

4.2.2.7 Petroleum Exploration and Development. Brines are almost universally associated with oil deposits. They are sometimes produced in greater quantities than crude oil, especially from older fields. Brine pits are the most common waste disposal facilities. In theory, water evaporates leaving salt accumulations. In practice, brine frequently leaks from the pit and carries high concentrations of salt into underlying aquifers. Sodium and calcium are the most common cations, with chloride, sulfate, and nitrate as the most common anions. Some brines contain enough bromide for economic recovery. Brine may also be used as a secondary recovery injectant. Indicative of the extent of the problem, over 400 billion gallons of brine were produced in the U. S. in 1974.[3]

4.2.2.8 Feedlots. Ground-water contamination from feedlots occurs principally from leaching of nitrate. Some mineralization of infiltrating water may also occur but not usually to an extent that serious contamination results.

Phosphate from feedlot runoff is a serious pollution hazard to surface water. However, phosphate is retained in soil and rarely moves into ground water. Of the heavy metals, zinc is present in the highest concentration in manure. None of the heavy metals are in concentrations as high as those associated with municipal sludges of mixed domestic and industrial origin. No appreciable contribution to ground water of manure-contaminated heavy metals is anticipated.

4.2.2.9 Waste Lagoons, Oxidation Ponds, and Buried Pipelines. Some sources of ground-water contamination are difficult to assign specific chemical constituents. Waste lagoons and oxidation ponds is one of these categories. Such facilities may contain a wide range of varying inorganic or organic elements or organisms. Therefore, in listing leachate indicators, probabilities are given of their occurrence in lagooned and ponded waste leakage as it enters the ground-water system (Table 8).

Buried pipelines and tanks are other sources of ground-water contamination. Probably the most common contaminants from these sources are petroleum products. Chemical storage tanks may also be included in this category. Again, the probabilities shown in Table 8 represent the probabilities of the given indicator actually reaching ground-water.

4.3 CHEMICAL, PHYSICAL, AND BIOLOGICAL INDICATORS

A comprehensive listing of leachate indicator parameters has been prepared and presented in Table 9. This listing reflects the most widely used leachate indicators by researchers in the field and state regulatory agencies.[4]

The schematic diagram on Figure 52 and the list of leachate indicator parameters in Table 9 represent the principal undesirable characteristics of leachate from municipal solid wastes. Its deleterious effects on ground and surface waters become apparent.

Some of the effects include:

- Dissolved oxygen depletion in surface waters caused by soluble organics and some inorganics;
- Objectionable tastes and odors in water supplies as a result of the presence of soluble constituents;
- Health hazards related to toxic materials and heavy metal ions and microbiological contaminants present in excess of drinking water standards;
- Limited use of ground and surface waters for drinking, domestic, industrial, or recreational use as a result of the effects of dissolved solids in excess concentrations.

These examples all point to the basic need for monitoring many of these parameters. Further discussion and background information regarding the undesirable nature and potential effects of the various leachate indicator parameters can be found in the Handbook for Monitoring Industrial Wastewater, U. S. EPA and in the introductory remarks of Standard Methods.[5,6]

The actual selection and use of indicators for a particular monitoring program will generally be decided from the indicators on Table 9 and will depend upon a number of considerations:

- Type of monitoring network;
- Susceptability to attentuation;
- Background-water quality;
- Location of well being sampled;
- Purpose of monitoring;
- Other considerations including cost, regulatory standards to be met, availability of laboratory equipment, availability of manpower, and simplicity and precision of determination;
- Type of solid waste handled and other site-specific factors.

4.4 INDICATOR GROUPS

Indicators can be categorized into various groups or levels of monitoring which vary in degree of information obtained in relationship to the purpose for the monitoring. Three such levels widely used are:

TABLE 8

PROBABILITIES OF LANDFILL LEACHATE INDICATORS FROM GIVEN SOURCES CONTAMINATING GROUND WATER

Indicator	Waste lagoons and ponds	Buried pipelines and tanks
Phosphate	II	III
Calcium	III	III
Magnesium	III	III
Sodium	I	II
Potassium	III	III
Ammonium	II	III
Chloride	I	II
Sulfate	I	II
Nitrate	I	II
Bicarbonate	III	III
Iron	I	III
Manganese	I	III
Boron	II	III
Selenium	II	III
Zinc	II	III
Copper	II	III
Lead	II	II
Other h.m.	II	II
MBAS	III	III
Phenols	I	I
PCB	II	III
Org N	II	III
PAH-HC	III	I
TOC	II	I
BOD	II	I
Coliform	III	III
Virus	III	III

I=Highly probable
II=Probable
III=Unlikely

TABLE 9

LEACHATE INDICATORS

Physical	Chemical		Biological
	ORGANIC	INORGANIC	
Appearance	Phenols	Total Bicarbonate Solids (TSS, TDS)	Biochemical Oxygen Demand (BOD)
pH	Chemical Oxygen Demand (COD)	Volatile Solids	Coliform Bacteria (Total, fecal; fecal streptococcus)
Oxidation-Reduction Potential	Total Organic Carbon (TOC)	Chloride	Standard Plate Count
Conductivity	Volatile Acids	Sulfate	
Color	Tannins, Lignins	Phosphate	
Turbidity	Organic-N	Alkalinity and Acidity	
Temperature	Ether Soluble (oil & grease)	Nitrate-N	
Odor	MBAS	Nitrite-N	
	Organic Functional Groups as Required	Ammonia-N	
	Chlorinated Hydrocarbons	Sodium	
		Potassium	
		Calcium	
		Magnesium	
		Hardness	
		Heavy Metals (Pb, Cu, Ni, Cr, Zn, Cd, Fe, Mn, Si, Hg, As, Se, Ba, Ag)	
		Cyanide	
		Fluoride	

- Specific Conductance Measurements;
- Key Indicator Analyses;
- Extended Indicator Analysis

4.4.1 Specific Conductance Measurements

When monitoring ground water and its temporal fluctuations, specific conductance is a useful parameter for approximating the total amount of inorganic dissolved solids. The real value of specific conductance as an indicator is that it can be measured quickly and easily, using inexpensive, accurate, and reliable portable equipment and requires a low level of training.

Specific conductance has been successfully correlated with total dissolved solids for monitoring leachate-enriched ground water. It also has been used successfully to detect fluctuations and trends for ionic impurities in the ground water. If conductivity is used as a quantitative indicator of total dissolved solids, it is absolutely essential that a valid correlation between the two parameters be determined for the specific land disposal site being monitored. Otherwise, gross errors can result in data interpretation. If however, specific conductance is used to make quality control comparisons or to check for leachate presence or absence, correlation with total dissolved solids is not as critical. For purposes of quality control, availability of a conductivity meter at a site would allow an operator to "spot check" monitoring wells at very frequent intervals, i.e. weekly or monthly. Where time and budget restrictions are a problem, specific conductance measurements can also be used advantageously during the sampling of a site to prioritize wells to be sampled.

4.4.2 Key Indicator Analyses Group

The intent of this monitoring group is to include highly sensitive analytical parameters which can be performed rapidly and accurately at a relatively low cost by personnel with a minimum of training to yield reliable, useful data. A selection should be made of a group of parameters which will provide information regarding ionic, nonionic, inorganic, organic, and suspended constituents of the ground-water sample. Most of the parameters should lend themselves to field analyses using portable equipment. Field analyses have the advantage of eliminating the necessity for low temperature and/or chemical preservation of the sample, thereby minimizing labor and deterioration effects on the analysis which can result from sample degradation due to aging. This monitoring group can be performed at frequent intervals with low cost, manpower, and equipment requirements.

The final selection of analytical parameters must consider the background-water quality, the pure leachate quality, as well as the hydrogeologic influences. The group must, therefore, be site specific as well as remain flexible to change at a site as may be dictated by data interpretation.

For discussion purposes, the following list of key indicators has been widely used in the field to determine the presence of leachate:

- Specific Conductance
- pH
- Temperature
- Chloride
- Iron
- Color
- Turbidity
- COD

This group fits well the criteria for key indicators and, with the exception of COD, can be performed rapidly using portable field equipment.

It is not suggested that all of the indicator parameters mentioned in this list must necessarily be used together to determine the presence of leachate. Rather, this is to be left to the judgment of the individual analyst. It is possible, for instance, that results from just one of the analyses (e.g., Specific Conductance) could indicate the probable presence or absence of leachate. A decision would then be made whether to run some or all of the remaining parameters, or additional tests to determine the reason for a high conductance value.

Data obtained from the key indicator analyses have value in and of themselves, i.e. individual determinations will give valuable information regarding the possible presence of leachate. In addition, data obtained from several indicator analyses can be cross-correlated and interpreted so that even more insight can be gained about the nature of the contamination, over and above what is obtained from the individual tests.

The following hypothetical examples serve to illustrate this point:

Example 1. A sample of ground water is analyzed and yields the following results: high levels of color and COD; low levels of iron, turbidity, and conductance. These results could be interpreted as an indication of the presence of an appreciable concentration of colored organic contamination in a system which is low in soluble and suspended inorganic contaminant levels.

Example 2. A sample of ground water is analyzed and yields the following results: high levels of conductance, chloride, pH, and turbidity; low levels of COD, color, and iron. These results could be interpreted as an indication of the presence of an appreciable concentration of inorganic materials, both suspended and in solution, and a low concentration level of organic materials.

Example 3. A sample of ground water is analyzed and yields the following results: high levels of conductance, chloride, iron, color, and COD; low levels of turbidity and pH. These results could be interpreted as an indication of the presence of appreciable concentrations of both inorganic and organic contaminants in acid solution and very low levels of suspended materials.

For a more positive affirmation of the presence of leachate, further interpretation of the indicator data must consider background water quality, hydrogeology, attenuation, and other pollution sources in the vicinity of the landfill.

4.4.3 Extended Indicator Analysis Group

This monitoring group is a much more comprehensive group of analytical parameters. Table 9 presents a comprehensive extended indicator analyses group which provides for a good characterization of the water samples and represents indicators commonly used by researchers and required by many regulatory agencies.[4] Performance of this monitoring group will obviously be costly, requiring trained personnel and an adequately equipped sanitary laboratory. Very few of the parameters can be analyzed with portable field equipment, thus requiring the utilization of acceptable storage and preservation techniques.

There can be a number of reasons for performing extended indicator analyses. The main reason to perform additional analyses is to provide more positive verification of contamination which has been indicated by the key indicator analyses group.

Additional testing, whether instituted by a regulatory agency or the landfill operator, should always be approached conservatively from both a technical and cost standpoint. Arbitrarily requiring an extended program without reasonable technical justification results in a very costly undertaking with little or no regard to its cost-benefit effectiveness.

An extended analysis program is justified when it can be demonstrated that a key indicator program does not have the necessary and sufficient capability to assure the absence of leachate contamination; to yield required background quality information; or to provide enough information to solve a specific contamination problem. When background quality information is being developed, a relatively large number of analytical parameters should be investigated in order to choose the few most valuable ones which will constitute the key indicator analysis program.

A sudden radical change in a key indicator may also point to the necessity for extended analytical work. For example, an elevated COD value might indicate contamination by a non-specific organic material. In this case, the indication would be to perform a number of analyses such as wet chemical tests or even infra-red and gas chromatography (if equipment is available), in order to determine which specific compound or compounds caused the change in the COD. From a regulatory standpoint, the nature of an extended analysis program would also be related to the standards which have been set to assure the absence of leachate contamination. For example, suppose that a state regulatory agency decides to adopt as its enforcement standard the U. S. Public Health Service Drinking Water Standards of 1962. This would result in an extended analysis program which would at least require testing for 24 parameters--4 physical and 20 chemical. As another example, a state regulatory agency may decide to adopt as its enforcement standards a series of seven physical and chemical parameters [e.g.: pH, specific conductance, chemical oxygen demand (COD), chloride, iron, color, and turbidity]. This program would per se constitute an adequate key indicator program, thereby eliminating the need for an extended analysis program for regulatory purposes.

Thus, it can be seen that the concept of an extended analysis program is a relative one. It is usually relative to regulatory requirements, the need for data in addition to the key indicator program, or the analysis of specific contamination problems. It must, therefore, be the task of the responsible engineers and analysts to determine what will constitute an extended analysis program, if any, for a given landfill site.

4.5 GUIDELINES FOR USING INDICATORS

For a given land disposal site, the selection and use of indicators will vary with the background-water quality, the differential attentuation that may occur and the monitoring well being sampled.

4.5.1 Background Water Quality Monitoring

4.5.1.1 New Land Disposal Site. For a new land disposal site, the background monitoring will define the naturally occurring constituents in the ground water and contaminants from other possible pollution sources that may be in the area. Section 4.2 presents a comprehensive summary of a variety of other pollution sources and related contaminants. Usually, the background quality at a new site can be satisfactorily defined by performing an extended indicator analysis group on an "A" well(s) that has been installed at the site for this purpose. Chapter 2 discusses various types of "A" wells in the different monitoring networks. It may be desirable to have additional "A" wells if other significant pollution sources are suspected or where more than one water-bearing zone is to be monitored.

For the initial sampling, the extended indicator analyses group should be comprehensive, such as the list of parameters of Table 9. Additional parameters may be deemed desirable where applicable to define "other pollution sources" in the area. In the case of the latter, the characteristics of other pollution sources should be investigated in selecting any additional parameters for monitoring. If possible, it is desirable to perform a few samplings (3 or 4) to obtain a statistically reliable data base for the long-term monitoring prior to commencement of operations. As is usually the case, time and economic considerations prohibit this pre-sampling procedure. Therefore, it is suggested that additional data be collected on a few selected key indicators which will likely be used in the long-term monitoring. These can be done quickly and cheaply and will provide valuable data in developing a statistically reliable data profile, as will be discussed later in this chapter.

While the "A" wells will establish the background water quality for the site, it is also important to develop background quality data for each "B" and "C" well. This is more important for larger landfills where the "A", "B", and "C" wells are relatively far apart in respect to changes in geology and other pollution sources which may influence their quality. Therefore it is recommended that an initial extended indicator analysis group be performed on all monitoring wells as soon as possible after their installation.

4.5.1.2 Existing Land Disposal Sites. For an existing land disposal site where solid waste has already been landfilled, the background-quality monitoring should acknowledge possible leachate contamination that may have already occurred on the site. Monitoring in this case must then involve the "A", "B", and "C" wells installed at the site. The "A" well will define the upgradient background quality while the "B" and "C" wells define the downgradient quality and any existing leachate contamination.

Here, the quality of the "B" well becomes especially significant in selecting the key and extended indicators for the monitoring program. This monitoring well will detect the leachate contaminants that are entering the saturated zone. The analysis of this well will also provide valuable information about the unknown past history of a site. For example, an old site may have previously disposed of hazardous wastes which could result in significant concentrations of exotic contaminants not normally attributed to municipal solid waste (e.g., certain heavy metals or pesticides). Depending upon the extent of the problem and the degree of potential hazard to the public, a decision can then be made as to the inclusion of additional parameters into the monitoring program for use as key or extended indicators. In any event, analysis of the "B" well will represent the "worse case" in terms of leachate contamination at a particular site and can provide a basis for including or excluding indicator parameters in the on-going monitoring program.

4.5.2 On-Going Monitoring. The on-going monitoring program should consist of the judicious use of representative key and extended indicator analyses groups--the former run at more frequent intervals and the latter run less frequently for verification purposes. The key indicator group is designed for the primary purpose of determining the presence or absence of leachate contamination and as a "check" on quality fluctuations. The extended indicator group is designed to provide verification of non-specific key indicators (e.g. COD or specific conductance) as may be required for purposes of enforcement, design, or scientific investigation.

4.5.2.1 Key Indicator Program. The on-going program will, of course, involve the monitoring of the "A", "B", and "C" wells. After establishing some background-quality data for the various wells (whether it be one or a series of samplings), the most representative indicators for the site would be selected. In doing this, one must consider the background-quality data, the constituents of the leachate, and the potential influence of attenuation. The information presented earlier in this chapter on background-water quality and in Chapter 5 on attenuation provides some valuable guidelines for selecting indicators.

As an example, suppose the natural background quality is high in iron or total dissolved solids. In this case, the high concentrations required to distinguish leachate from background water quality lessens the value of iron and specific conductance as key indicators of leachate contamination. Or, there may be another pollution source in the vicinity that is affecting the background quality of the ground water, such as: deicing of adjacent highways, seepage from local septic tanks, or leaky sewers. These, too, might serve to lessen the value of various parameters as leachate indicators.

Susceptibility of attentuation in different soils would also affect the value of the various parameters as incidators of leachate. The information presented in Section 5.2 and Table 12 can be used as a guide in indicator selection for a particular disposal site. Because of its freedom from attenuation, chloride is noted as a popular and useful indicator. As long as no presence of leachate is detected or no significant fluctuations in the data are observed where leachate is already present, the key indicator program should be followed. Another consideration, however, would be the regulatory agency's requirements. Some states do require a periodic (e.g., semi-annual or annual) testing for an extended analysis group regardless of the monitoring trends being observed.

In most cases, all "A", "B", and "C" wells at the site should be included in the key indicator program. However, this is not to be considered an ironclad rule. Many large acreage land disposal sites may have a large number of wells (20 to 30). In these cases, one may elect not to sample all wells at each sampling, but to rotate sampling to include a small number of wells each sampling date. Therefore, in the case of quarterly sampling, each well would be sampled once per year. Generally, each sampling should include at least one of each well type, i.e., an "A", "B", and "C" well. The convenience of the specific conductance test can be a valuable asset in this case. The ability to be tested quickly with a field instrument may allow at least a specific conductance reading on all wells at each sampling date. The specific conductance reading provides the added dimension of making on-the-spot decisions as to which wells should receive sampling priority. Having the specific conductance data profile on hand for quick reference, the field technician can compare the most recent reading with the profile. A significant change worthy of further investigation may be indicated. A combination of a routine rotational sampling schedule that is subject to possible modification due to significant changes in specific conductance would comprise a sound rationale to the use of the indicator group.

If a significant change is observed in a key indicator parameter or parameters (e.g., increase in specific conductance), the possibility of leachate contamination should be suspected and a plan for further investigative and corrective action should be instituted immediately. This plan should include additional sampling and analytical work as determined by the key indicator data obtained. This data should serve as a base for developing a satisfactory corrective action to eliminate the cause of the contamination problem and to monitor the effect of implementing the same.

Assuming that a leachate contamination problem has been discovered by the key indicator program and corrective action implemented, sampling and key indicator testing should be instituted on an increased level of frequency, with the possible inclusion of additional parameters. This should be continued until there is reasonable certitude that conditions have returned to normal and will probably remain so. At this point, a re-assessment of the key indicator analytical program should be made. It should then be decided whether to reinstitute the program in its original form or to initiate it in a modified form based upon experience gained in solving the contamination problem.

The following hypothetical example will serve to illustrate this approach:

> Sulfate was found to be a principal contributing contaminant and had caused a high specific conductance reading. It was then decided to test for sulfate, for a limited time, in addition to the other parameters of the key indicator program. In this manner, valuable data was collected that showed sulfate/conductance ratios which could be used as a guide in monitoring for future problems.

4.5.5.2 Extended Indicator Program. Obtaining background-quality data and further investigating a particular problem are the principal technical reasons for implementing an extended indicator analyses group. In the legal sense, enforcement data needs may require extended indicator analyses data. Administratively, many regulatory agencies will require sampling for an extended indicator analyses group. The extended indicator program should be used judiciously due to the relatively excessive costs and manpower requirements associated with their performance.

The fact that the extended indicator parameters are basically serving to verify the results of the key indicator parameters should provide a basis of a rationale for selecting and ranking the former. In other words, a significant fluctuation in a particular key indicator would "trigger" further investigation by a select group of extended parameters and not automatically the entire extended indicator analyses group. The implementation of the extended indicator analyses group should also be a monitoring well specific decision. For example, a quality change in one "C" well should not immediately require the testing for extended indicators in all of the "C" well.

The following examples of relationships between key and extended indicators serve to illustrate the point:

- A significant change in specific conductance would be an indicator of possible changes in levels of one or more of the following extended indicator parameters: pH, total dissolved solids, chloride, sulfate, phosphate, alkalinity, acidity, nitrogen series, sodium, potassium, calcium, magnesium, hardness, heavy metals, cyanide, fluoride, and COD.

- A significant change in chloride concentration would be an indicator of possible changes in levels of one or more of the following extended indicator parameters: specific conductance, total dissolved solids, pH, acidity, and metal ions.
- A significant change in iron (total) concentration would be an indicator of possible changes in levels of one or more of the following extended indicator parameters: specific conductance, pH, total dissolved solids, chloride, sulfate, phosphate, manganese, and fluoride.
- A significant change in color would be an indicator of possible changes in levels of one or more of the following extended indicator parameters: COD, TOC, tannins, lignins, organic N, total dissolved solids, pH, iron, BOD, and conductance.
- A significant change in turbidity would be an indicator of possible changes in levels of one or more of the following extended indicator parameters: pH, conductance, COD, TOC, tannins, lignins, total suspended solids, phosphate, alkalinity, acidity, calcium, magnesium, hardness, heavy metals, fluoride, and BOD.
- A significant change in COD would be an indicator of possible changes in levels of one or more of the following extended parameters: BOD, pH, conductance, TOC, volatile acids, tannins, lignins, organic N, total dissolved solids, total suspended solids, volatile solids.

In a practical situation, several of the key indicators will most probably show variations at the same time. Therefore, looking at combinations of key indicators will provide additional information for the analyst to define the chemistry of the system involved and further assist in specifying additional extended indicators for analysis.

4.6 MONITORING FREQENCY

The sampling schedule for a land disposal site should be flexible, allowing for modification. Monitoring frequency is greatly influenced by many factors, such as:

- Characteristics of ground-water flow;
- The location and purpose of the particular monitoring well;
- Trends in the monitoring data;
- Legal and institutional data needs;
- Climatological characteristics;
- Other considerations.

The hazardous nature of the leachate in relationship to the threatened water resource (e.g., a single domestic well or an entire municipal water supply) will to a large degree dictate the monitoring effort.

4.6.1 Characteristics of Ground-Water Flow

The principal characteristics of concern in selecting a sampling frequency is the rate of ground-water flow at the land disposal site. The flow rate will be primarily dependent upon the aquifer porosity and permeability as well as the hydraulic gradient existing at the site. Aquifers are generally categorized by porosity into intergranular porosity, fracture porosity, and solution porosity. Ground-water flow rates ranging in orders of magnitude from a few meters (feet) per year in an impervious intergranular porosity aquifer to tens of meters (feet) per day in the more unpredictable fracture and solution porosity aquifers (Chapter 2).

The higher the rate of ground-water flow, the greater the monitoring frequency needed. Two extreme examples would be an intergranular porosity aquifer with impervious clay soils vs. a fracture or solution porosity aquifer with unpredictable and high flow rates likely. Hypothetically then, if the closest "C" well is 30 meters (100 feet) from the landfill and the closest downgradient property line or domestic well is 90 meters (300 feet) away, at the site with clay soils it would be senseless to do frequent sampling since, theoretically, it would take ten to twenty years for any leachate-enriched ground water to even reach the well. In this instance after establishing background quality, a semi-annual or annual monitoring of the well with select key indicator parameters would suffice. In the case of a fracture or solution porosity aquifer, it is possible that contaminants could migrate from the property in a matter of days, weeks, or months. Here, a quarterly monitoring with key indicators and a more frequent spot-checking with specific conductance would be warranted. As was previously discussed, the extended indicator group would be utilized as needed. Most landfills will fall between these two extremes; however, careful consideration must be given to the flow rate and the distance involved to select a frequency which will not miss an environmental occurence.

In a similar sense, the monitoring well would be influenced by the vertical flow rate of leachate-enriched ground water. For example, if a disposal site is underlain by a sand aquifer with a relatively high ground-water flow rate and is separated from the landfill by a thick layer of impervious clay, a concentrated monitoring effort of the "C" wells in the aquifer would not be justified until the "B" well detected that contaminants have traveled through the clay layer and reached the aquifer.

4.6.2 Location and Purpose of the Monitoring Well

The distance that a monitoring well is located from the land disposal site as well as its depth will influence monitoring frequency. For example, there may be a case where a line of "C" wells are placed along the property line for legal and administrative reasons due to ground-water protection laws. There is little need to concentrate on monitoring these wells until the monitoring results of closer "C" wells present some reason to believe that leachate contaminants may be approaching close to the property line. Only minimum monitoring of these wells to establish background quality and meet regulatory requirements would be justified.

Another example might be a well located in deep water bearing zones separated from the disposal site by other aquifers and aquicludes. Chapter 2 depicts examples of areas (e.g., coastal plain) where there are a series of alternating aquifers and aquicludes. For institutional reasons or regional water planning purposes, a monitoring well may be placed in a deep aquifer which has almost no chance of being contaminated by the land disposal site. After an initial monitoring for background purposes such wells would only deserve less frequent attention--unless, of course, other monitoring results cause reason for concern.

4.6.3 Climatological Characteristics

In setting up the initial monitoring schedule for a particular site, the fluctuations in leachate generation that occur over the year should be analyzed. The water balance method (Chapter 5) is a very useful tool for this purpose. As an example, it may be desired to perform quarterly sampling. Instead of arbitrarily assigning a sampling date every third month, the monitoring schedule should be correlated with ground-water recharge periods in the year when leachate generation is greatest. Selection of the actual sampling dates should also take into account the well location and depth, ground-water flow rate, saturation condition of the landfill, and other factors to project approximate lag times that may occur between first appearance of leachate and its impact on the monitoring well.

4.6.4 Trends in the Monitoring Data

The three factors presented above (ground-water flow rate, well purpose and location, and climate) will be used at the outset in establishing monitoring frequencies. However, monitoring frequencies should never be considered ironclad. They should maintain flexibility for modification to respond to trends in the monitoring data.

As an example, if a spot check with a specific conductance meter indicates a significant change in the water quality at a particular well, further investigation with additional key and extended indicators would be desired immediately, regardless of the next sampling schedule. Concentration on this well might also reduce the frequency at another well where recent data has shown no significant changes in water quality.

4.6.5 Legal and Institutional Data Needs

Monitoring frequencies at a site may also be altered for legal and institutional reasons. As an example, if an enforcement action is initiated against a landfill, to strengthen their case, attorneys for both the State and the disposal site may request that all of the monitoring wells be monitored for an extended indicator analyses group.

4.6.6 Other Considerations.

Other reasons for modifying the monitoring frequencies at a site would include:

- Complaints from neighboring residents;
- An unusually severe climatological event such as a hurricane, with large amounts of rain in a short time period;
- A sudden change in or addition of an "other pollution source", such as an oil spill adjacent to the property;
- An unusual operational occurence, such as the illegal and/or improper dumping of a large volume of liquids at a site.

A properly planned monitoring program will allow for modification in sampling schedules to respond to the above-mentioned occurrences.

4.7 COST CONSIDERATIONS

In selecting indicator parameters and sampling frequencies, it is important to be mindful of relative costs for performing the monitoring. The three basic levels of indicators used for monitoring presented earlier vary not only in the depth of analytical data provided but also in the costs for sampling and analysis.

Specific conductance is valuable because it is so inexpensive to perform. By using a portable field meter, the analytical cost is merely the few extra minutes required by the technician to perform the test at the site. The meter itself is relatively inexpensive, costing approximately $300.00 (1976 prices). The sampling costs are also low, primarily because it is not necessary to collect and preserve samples for the laboratory. This lessens the amount of bottles to be carried and the time for adding sample preservatives. This advantage would be somewhat limited where pre-pumping of the wells is done. There is no way of estimating sampling costs and time requirements since they are site specific depending upon accessibility and number of wells as well as the pre-pumping and sample withdrawal method used. For order of magnitude comparison purposes only, a typical commercial laboratory would charge approximately $3.00 per sample for a specific conductance analysis (New York Area, 1976 prices), exclusive of sampling.

A typical key indicator analysis group, i.e. specific conductance, pH, temperature, chloride, iron, color, turbidity, and COD would be more expensive for sampling and analysis. With the exception of COD, all the analyses can be run in the field with portable equipment. Sampling time will be increased, however, by the additional equipment and analyses required and the time to collect, store, and preserve the COD sample. Where one man could manage nicely with specific conductance measurements, an assistant may be desirable in monitoring for all the key indicators depending upon the number and accessibility of wells to be sampled and the sample withdrawl method used. Adverse weather conditions may also necessitate transporting samples back to the laboratory for analyses where as specific conductance could still be done in the field. For order of magnitude comparison purposes, a typical commercial laboratory would charge approximately $50.00 per sample for the key indicator analyses listed (New York Area, 1976 prices), exclusive of sampling.

A typical extended indicator analysis group, such as listed in Table 9, would be the most expensive level of monitoring for both sampling and analysis. With the exception of some key indicators which might be run in the field, all the indicators require proper storage, preservation, and transport of samples to the laboratory for analysis. This would require additional sampling time and possibly additional manpower to sample properly and efficiently in the field. Of course, adverse weather conditions may further complicate sampling efforts.

Again, for order of magnitude comparison purposes, a typical commercial laboratory would charge approximately \$600 - \$700 per sample for the extended indicator parameters listed in Table 9, exclusive of sampling.

Table 10 summarizes the cost of analysis for the various monitoring groups discussed above. As noted, no comparisons of sampling costs have been made due to its extreme site specificity.

4.8 DATA MANAGEMENT

4.8.1 General

At a given land disposal site, appreciable quantities of data relative to ground-water quality will be generated over a period of time. Several factors govern the amount of data produced, including the number of monitoring wells, the number of parameters to be tested, and the frequency of testing (both scheduled and unscheduled).

As a hypothetical case, assume that there are 20 monitoring wells at a given landfill and that the following tests are performed in a given year:

Testing Category	No. of Parameters	No. of Wells	No. of Tests
Annual-Extended	30	20	600
Quarterly-Indicator	10	20	600 (200 x 3)
Problem-Unscheduled	30	20	600
		Total	1,800

The total number of tests performed in the landfill over a period of one year will be 1,800. This figure could approach 20,000 over a 10-year period. This amount of raw data must be processed, interrelated, statistically analyzed, and stored in readily retrievable form so that it will be of maximum value for quality control, engineering, and legal purposes.

The use of digital computer treatment would appear to be an excellent tool; it is both rapid and cost-effective as a management

TABLE 10

COMPARATIVE COSTS OF INDICATOR ANALYSES*

Monitoring Group	Approximate Cost of Analysis+ Per Sample ($)
Specific Conductance	$ 3.00
Key Indicators	$50.00
Extended Indicators	$600.00-$700.00

Note: It should be noted that there is an economy of numbers relative to both sampling and analysis. Appreciable quantity discounts are usually available for different levels of sampling and analysis. Additional savings can usually be realized through the use of a long-term sampling and analysis contract.

*A comparison of sampling costs has not been made due to its extreme site specificity - such a comparison should consider the number and accessibility of wells, weather conditions, whether or not the wells will be pre-pumped prior to sampling, and the pre-pumping and sample withdrawal methods used.

+Based on January 1976, rates of a typical commercial laboratory in the New York Area. Refer to Chapter 7 for laboratory manpower requirements for analyses.

information system in the handling of this type of data. Other approaches to data management could entail manual processing, storage, and retrieval of the data in the form of tables, charts, and graphs which can show parameter levels and trends relative to standard values. In both cases, the statistical handling and use of analytical data for quality control purposes in the form of ranges, means, standard deviations, parameter ratios (e.g., specific conductance/TDS ratio), and control charts will be important.

The technological state of the art of land disposal is still relatively young and is highly dependent upon monitoring for its development. Even if a particular design and operational strategy is successful at one site, it cannot be automatically assumed acceptable for all sites due to the extreme site specificity which is fundamental to land disposal of solid waste. Again, monitoring becomes critically important. Thus, the parameters monitored and the significant results obtained from the monitoring program will be critically evaluated in assessing a site and related design and operational approaches and in deciding upon modification. Because of the significance which may be placed on the results of the monitoring program, it should be the desire of the landfill management to understand and attempt to identify the causes of fluctuations in monitoring data obtained. Incorrect interpretation of monitoring data may result in unnecessary expenditures or in a false sense of security.

The variability of the indicator parameters measured in a monitoring program may result from various phenomena, some of which are:

- Natural fluctuations in the background-water quality;
- Occurrence of another pollution source which might cause the background-water quality to fluctuate;
- Attenuation taking place in the subsurface environment;
- Climatological variations;
- Operational deficiencies, incidents, and modifications;
- Experimental errors in the analyses of measured parameters;
- The sampling method utilized.

Variations in the background-water quality will occur with location and time. Such variations recorded in the "A" wells should be fingerprinted statistically to allow for more accurate interpretation of the data fluctuations recorded at the "B" and "C" wells. In the same vein, fluctuations in background quality may be artificially induced by another pollution source. As was stressed earlier, such occurrences must be carefully recorded because of their effect on monitoring data interpretation. Differential attentuation (Chapter 5) and climatological variations (Section 4.6.3) will have a definite influence on temporal and distance variations in monitoring data.

Operational factors will have a definite influence on the monitoring data and its evaluation and should be carefully documented. For example, operation changes, such as: type of wastes, a sudden disposal of a large quantity of liquid wastes, a deficiency in cover, or construction of the dikes, diversion ditches etc...could all significantly affect monitoring results and should be carefully described with dates recorded.

The results of the analysis of a sample by the same or different technicians using the same laboratory techniques often fluctuate widely. Even very accurate laboratory analysis cannot prevent a relatively wide range in determined values of parameters such as BOD, which may experience experimental error as high as ±20%. Variations will also exist with alternate analytical methods, especially field versus laboratory methods. This becomes even more significant for concentrated leachate samples where interferences further complicate analysis. Because of its significance in data interpretation, all of this information must be carefully recorded.

Because of the variations that can be created, differences in the sampling withdrawl method utilized is important for monitoring data evaluation. Questions that should be asked are:

- Was the well flushed out prior to obtaining a sample?
- Was the sample collected aerobically or anaerobically?
- Was the sample properly preserved?
- How much time ellapsed between sample collection and analysis?
- Who did the sampling?

It is important that these questions be answered and their implications understood when evaluating the monitoring results.

Therefore, all of the possible causes in variations should be carefully recorded and identified for proper evaluation of the monitoring results. It will be important that the monitoring program distinguish between fluctuations which are significant and attributable to the landfill requiring some form of remedial action versus those variations which are insignificant or not attributable to deficiencies at the landfill. A complicating feature for a land disposal site, unlike water and air pollution, is the time lag which inherently exists between cause and effect. For example, it may take months or years for a fluctuation observed in an "A" or "B" well to reach a distant "C" well thus often further complicating data interpretation.

4.8.2 Application of Statistics

In the Handbook for Monitoring Industrial Wastewater, the value of statistics in monitoring is discussed:[5]

> Statistics aid in the development of general laws resulting from numerous individual determinations which, by themselves may be meaningless. The resulting relationships are part of the fundamental function of statistics which expresses the data obtained from an investigative process in a condensed and meaningful form. Thus, the average or mean is often used as a single value to represent a group of data. The variability of the group of observations is expressed by the value of the standard deviation and trends in concentrations during the monitoring process are expressed in the form of regression coefficients.
>
> In general, the concern is with the treatment of the collected data. The accuracy or usefulness of these data is greatly enchanced if a full understanding was involved in generating the facts. The balance between use of statistical methods and evaluation based upon physical understanding is extremely important. The use and value of statistics decreases as physical understanding increases. Specifically, the difficulty lies in separating chance effects from valid occurrences. With the knowledge of basic probability theory and the use of statistical techniques, such as Least Squares Curve Fitting, Analysis of Variance, Regressive and Correlation Analysis, Chi-Squared Goodness of Fit, and others, it is possible to construct mathematical models and curves of almost any level of precision desired. Such techniques help to evaluate in information having wide variations, so that an estimate of the best value of the parameter being measured can be assigned; and also to assess the precision of that estimate. Statistical procedures may also help in identifying errors and mistakes and are helpful in comparing sampling methods and procedures and in evaluating waste loadings from different process schemes.

Because of the extreme site specificity of the various phenomena involved, evaluation based upon physical understanding (Section 4.8.1) is especially significant for monitoring a land disposal site.

Probably, the major use of statistics in a monitoring program is to correlate the data for the proper choice of statistical parameters (mean, range, and standard deviation) for the specific indicators for evaluation and comparison purposes.

Statistics and data analyses are very broad topics and are beyond the scope of this manual. The aforementioned EPA publication, Handbook for Monitoring Industrial Wastewater, cites several good references on statistics. It should be emphasized that rules and formulas for data analyses are many. To be of value, they must be chosen wisely and applied correctly.

4.8.3 Indicator Data Profiles

Once a monitoring program has been in operation for an appreciable period of time, the data obtained from it can be used to provide specific analytical profiles for ground water and/or surface water for a given land disposal site. These profiles will be characterized by data from a number of sampling points within the landfill and will reflect the influence of the various phenomena that result in fluctuations in the indicator parameters. Statistical analyses of the profiles will provide such important statistical values as normal ranges, means, and standard deviations for each of the indicator parameters.

Quality control data of the landfill site can be obtained from the profile data. This could take the form of control charts for the various parameters which would indicate whether the operation was "in control" or "out of control" relative to upper and lower control limits provided by the control chart.

Statistics can play an important role in the correlation of specific parameters, especially in the case of specific conductance to other parameters such as total dissolved solids. The Handbook for Monitoring Industrial Wastewater,[5] presents an excellent discussion on the statistics for correlation of specific parameters. The data profile will also provide an insight into the interrelationships of the various key indicator parameters in the form of normal ratios, (e.g., specific conductance to total dissolved solids) which should be developed for a cost effective monitoring program. When enough data are obtained on indicator parameter ratios for a given landfill site, statistical values of range, mean, and standard deviation can be developed as is done for the individual indicator parameters themselves. This information can be used as a valuable statistical tool for quality control of the landfill and as an aid in the diagnosis of leachate contamination problems and their probable causes.

In a monitoring program, data profiles can be used in a variety of ways, some of which are:

- Concentration of the various indicator parameters vs. time for each monitoring well. This is perhaps the most common use of a data profile in monitoring programs. It provides an immediate visual picture of the trends in quality and is nicely defined with the basic statistical values (mean, range, and standard deviation). It provides a valuable tool in comparing monitoring results of the "A", "B", and "C" wells for operational and enforcement purposes. It provides a readily available and convenient tool for comparing water quality trends to trends and occurrences in the various phenomena that influence the ground-water quality which were discussed earlier.

- Concentration of the various indicator parameters versus distance from the landfill. This type of profile would be constructed by plotting the data for selected indicator parameters which are obtained on a particular date for the various "C" wells located at different distances from the landfill. The quality of the "B" well would represent the concentration at zero distance from the landfill. Chapter 5 discusses the use of this type of profile in the measurement of attentuation at a land disposal site. Figure 54 shows an example of this profile.

- Maps with contour lines of equal concentration for key indicators can be prepared for the land disposal site. This technique is often used to present the results of resistivity surveys. Such a map has a value in outlining the aerial dimensions of a leachate plume and, if done periodically, trace the movement of contaminants. The U. S. Geologic Survey has prepared an excellent paper on this and other methods of ground-water data interpretation and presentation.[7]

- Other profiles providing "physical understanding" information. In the evaluation of the monitoring data obtained at a land disposal site, it would be of value to be readily accessible to information on the phenomena which may be the potential cause of fluctuations and trends in the monitoring data. This might include a water-balance profile showing ground-water recharge periods of a "chronological events" profile of important occurrences. A water

> balance profile, such as shown in Figure 53 could be developed for each site as part of the permit applications or for several representative "typical sites" throughout the state. The actual quantities are not as important as the trends they would depict. A profile of "chronological events" could be kept on file and easily up-dated by the inspection and monitoring personnel. Reference to such a profile would be of value, providing physical understanding in evaluating monitoring results, consideration should be made for the time lag between cause and effect that is inherent at land disposal sites.

An indicator program based upon sufficient background quality data and on-going statistical information should provide a basic, cost-effective, reliable monitoring tool for the quality control of a landfill.

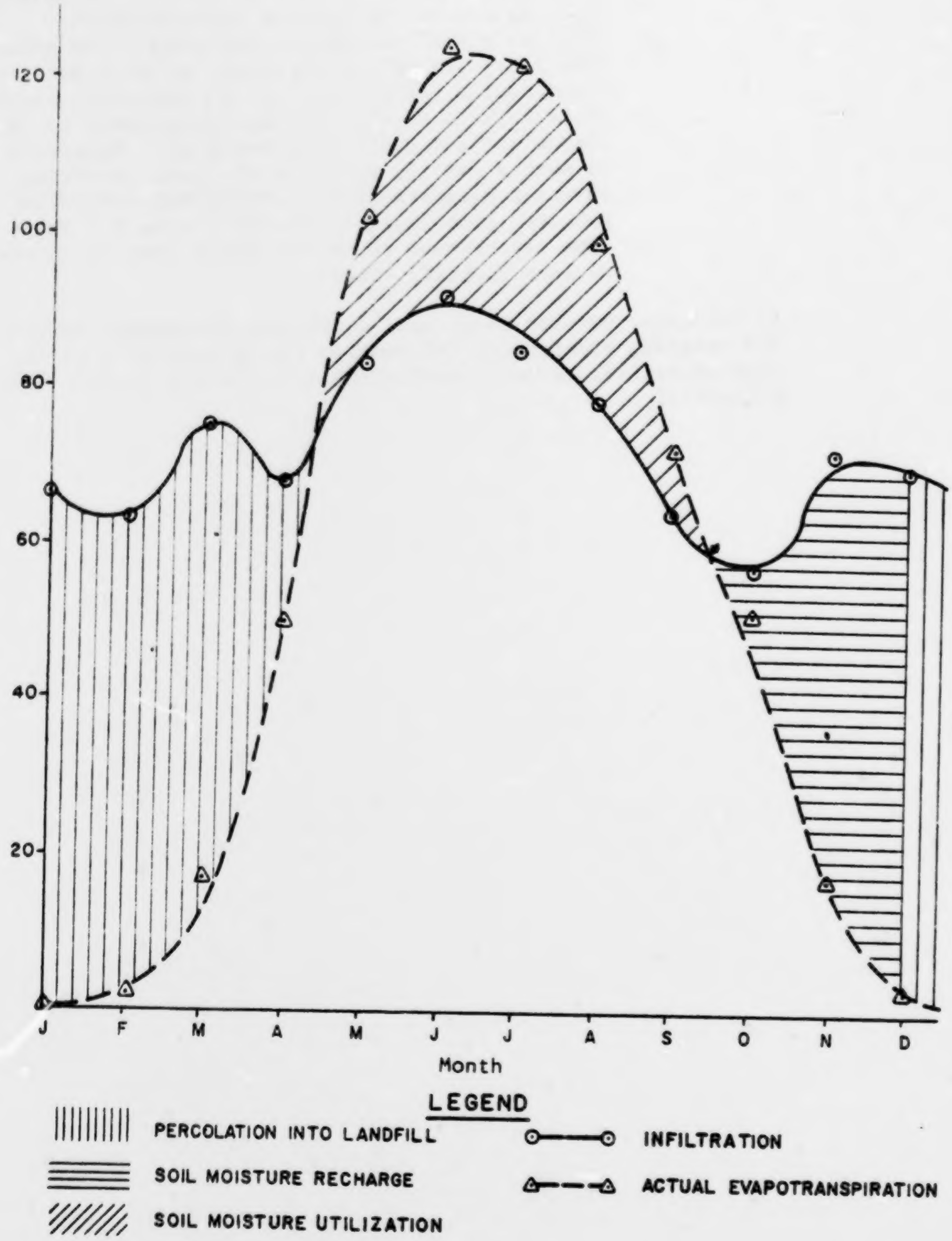

FIGURE 53. WATER BALANCE PROFILE
(Fenn, et. al., 1975)[6]

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5.0 FUNDAMENTALS OF LEACHATE

5.1 INTRODUCTION

As discussed in Chapter 1, it is important to understand and assess the potential for leachate contamination at a land disposal site in order to properly design, implement, and interpret a monitoring program and its data. Of concern here is leachate production: its quality, quantity, and its fate in the hydrogeologic environment. A clear understanding of each of these concepts, their underlying theories, causes, and results should be prerequisite to the design and implementation of a monitoring program.

An overview of the fundamentals of leachate is presented in this chapter for use in assessing the potential for leachate contamination at a land disposal site. If properly applied, the resultant information will be useful in the design and implementation of a monitoring program and interpretation of the resultant data. Further, the information presented may be useful to regulatory officials in preparing background and reference material for enforcement actions.

In approaching the monitoring of a land disposal site, the following questions should be asked:

- For what kind of contamination is the landfill being monitored?

- How much contamination in terms of concentration and quantity can be expected?

- Where, how fast, and how far will the contamination travel?

- What is the best monitoring method for the contamination?

Addressing these questions requires a clear understanding of leachate production and its fate in the landfill and surrounding environment.

5.2 ORIGIN, COMPOSITION, AND FATE OF LEACHATE

In understanding the quality of leachate contaminants and the concentrations that may be encountered by monitoring, consideration should be made as to changes in its quality as it emanates from the compacted solid waste and travels in the subsurface environment.

The former would be the quality of pure leachate, while the latter deals with the quality of leachate-enriched ground water.

Where precipitation is greater than runoff and evapotranspiration combined, leachate moves downward through solid waste and underlying soil and sediment until it reaches an impermeable layer or ground water. In its journey leachate traverses three zones of geochemical activity with certain characteristics which are shared and others which are unique. This section describes some of the characteristics in each of the zones and explores the ways they affect the mobility of leachate.

5.2.1. Solid Waste Zone

Solid waste deposited in municipal landfills is a heterogeneous mixture of organic and inorganic materials and living organisms. Upon deposition, and frequently before, microbial activity begins the degradative process on organic matter. The microbial decomposition of organic matter is encouraged by moisture and warm temperatures.

Microbial activity soon depletes the oxygen supply and causes the solid waste beyond the zone of rapid air diffusion to go anaerobic (absence of oxygen). Ananerobic conditions cause the end products of decomposition to be somewhat different from carbon dioxide and water (the products of complete oxidation), notably methane gas. Other organic anaerobic decomposition products such as alcohols, aldehydes, and thiols tend to be more odoriferous than their aerobic counterparts. Of particular importance with regard to leachate are the anaerobic forms of sulfur, nitrogen, iron and manganese.

The percolate flows downward through the solid waste which is in progressively advanced stages of decomposition. It then passes through layers of buried cover material. Percolate shows a net gain in dissolved constituents as it progresses downward but may lose some individual ions from cation exchange or other reactions encountered en route.

Nitrogen present in solid waste organic matter is released in soluble form with microbial decomposition. In organic substances, nitrogen is in a chemically reduced state. With aerobic decomposition, the nitrogen is oxidized to nitrate ion. Under anaerobic conditions, nitrogen is released as the ammonium ion. Since anaerobic conditions are predominant in landfills, most nitrogen in leachate is present as ammonium. The relatively small amount of nitrate produced, coupled with its probable denitrification, explains the typically low nitrate concentrations in leachate.

Aerobic decomposition of the organic matter in solid waste releases carbon dioxide in larger amounts than under anaerobic conditions.

The enrichment of the interstitial gas in solid waste by carbon dioxide results in production of bicarbonate ion which is frequently a major anion in leachate. Because of the reversibility of the reaction producing bicarbonate, it acts as a pH buffer.

Heavy metals in landfills are primarily in their metallic state and are not soluble. The exception is with deposition of soluble heavy metal salts either as solids or in solution. These may come from certain industrial activities such as electroplating or metal pickling. Most heavy metals occur in solution as cations, but a few are usually present as anions. Anionic heavy metals include vanadium, chromium, and molybdenum. Phosphorus is released to percolating water by decomposition of organic matter. Soils have a high capacity for phosphate attenuation as oppossed to solid waste material which does not. Phosphate can be, and frequently is, produced in substantial amounts of leachate. Were leachate to enter ground water directly, it would almost certainly contribute more phosphate than would leachate which has passed through soil.

Water quality parameters which do not measure individual chemical species include biochemical oxygen demand (BOD), chemical oxygen demand (COD), total organic carbon (TOC), color, conductance, and turbidity. The solid waste zone provides little, if any, attenuation of these parameters--instead it usually contributes to them.

Fecal coliform and fecal streptococci have been observed in leachate, and poliovirus was reported in leachate from a simulated landfill. The recent trend in the use of disposable diapers has increased the source of enteric bacteria and viruses in solid waste. Sewage sludge and septic tank pumpings, a principal source of bacteria, are also frequently disposed of in municipal landfills.

Movement of bacteria and viruses within the landfill and through the unsaturated zone is dependent upon the porosity of solid waste and underlying geologic formations. Solid waste may offer many paths through which water can travel relatively unimpeded. If coarse sand and gravel or fractured rock underlie the solid waste, percolating water may carry microorganisms with little or no attenuation except for natural die off. The latter conditions, judging from locations which have been studied, are the exception rather than the rule.

Much data on leachate quality has been reported in literature and is worthy of note. The U.S. Environmental Protection Agency has prepared Table 11 which illustrates some of the chemical and biological characteristics found in leachate and compares fresh leachate to a typical domestic waste water. The quality of leachate depends upon many variables which are specific to each land disposal site. Therefore, a recent Environmental Protection Agency report emphasizes the cautious interpretation of reported leachate data: 1

TABLE 11

CHARACTERISTICS OF LEACHATE AND DOMESTIC WASTE WATERS

Constituent	Range* (mg/l)	Range + (mg/l)	Range‡ (mg/l)	Leachate§ Fresh	Leachate§ old	Waste water§	Ratio§
Chloride (Cl)	34-2,800	100-2,400	600-800	742	197	50	15
Iron (Fe)	0.2-5,500	200-1,700	210-325	500	1.5	0.1	5,000
Manganese (Mn)	.06-1,400	--	75-125	49.	--	0.1	490
Zinc (Zn)	0-1,000	1-135	10-30	45	0.16	--	--
Magnesium (Mg)	16.5-15,600	--	160-250	277	81	30	9
Calcium (Ca)	5-4,080	--	900-1,700	2,136	254	50	43
Potassium (K)	2.8-3,770	--	295-310	--	--	--	--
Sodium (Na)	0-7,700	100-3,800	450-500	--	--	--	--
Phosphate (P)	0-154	5-130	--	7.35	4.96	10	0.7
Copper (Cu)	0-9.9	--	0.5	0.5	0.1	--	--
Lead (Pb)	0-5.0	--	1.6	--	--	--	--
Cadmium (Cd)	--	--	0.4	--	--	--	--
Sulfate (SO_4)	1-1,826	25-500	400-650	--	--	--	--
Total N	0-1,416	20-500	--	989	7.51	40	25
Conductivity (µmhos)	--	--	6,000-9,000	9,200	1,400	700	13
TDS	0-42,276	--	10,000-14,000	12,620	1,144	--	--
TSS	6-2,685	--	100-700	327	266	200	1.6
pH	3.7-8.5	4.0-8.5	5.2-6.4	5.2	7.3	8.0	--
Alk as $CaCO_3$	0-20,850	--	800-4,000	--	--	--	--
Hardness tot.	0-22,800	200-5,250	3,500-5,000	--	--	--	--
BOD_5	9-54,610	--	7,500-10,000	14,950	--	200	75
COD	0-89,520	100-51,000	16,000-22,000	22,650	81	500	45

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*Office of Solid Waste Management Programs, Hazardous Waste Management Division. An environmental assessment of potential gas and leachate problems at land disposal sites. Environmental Protection Publiciation SW-110 of. [Cincinnati], U.S. Environmental Protection Agency, 1973. 33 p. [Open-file report, restricted distribution.]

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‡Gas and leachate from land disposal of municipal solid waste; summary report. Cincinnati, U.S. Environmental Protection Agency, Municipal Environmental Research Laboratory, 1975. (In preparation.)

§Brunner, D. R., and R. A. Carnes. Characteristics of percolate of solid and hazardous waste deposits. Presented at AWWA American Water Works Association 94th Annual Conference, June 17, 1974. Boston, Massachuetes. 23 p.

> The compositions of leachates reported in the literature are quite diverse... The breadth of reported data are also typical for individual studies over a long period of time. The many factors that contribute to the spread of data are time since deposition of the solid waste; the moisture regimen, such as total volume, distribution, intensity, and duration; solid waste characteristics; temperature; and sampling and analytical methods. Other factors such as landfill geometry and interaction of leachate with its environment prior to sample collection also contribute to the spread of data. Most of these factors are rarely defined in the literature, making interpretation and comparison with other studies difficult, if not rather arbitrary. (sic)

In this same report, the Environmental Protection Agency has prepared a comprehensive summary of quality data for both pure leachate and leachate-enriched ground water as has been observed and reported by many researchers. In addition, the report addresses the significance of microbiological organisms in solid waste.

5.2.2 Unsaturated Zone.

Pertaining to this chapter, the unsaturated zone is defined as the area in soil or sediments between the bottom of the landfill deposits and the water table. The distance can vary between zero (solid waste contacting ground water) to hundreds of meters (feet). The zone is below what is usually considered "topsoil" or weathered, organic-matter-rich upper horizons of most soils. At most landfill sites, topsoil has been removed (sometimes much subsoil, also) prior to deposition of solid waste. The porous materials comprising the subsoil are likely to be low in organic matter, have a sparse microbial population, and may vary widely in permeability. For purposes of discussion, the unsaturated zone will be considered at 6 to 60 meters (20 to 200 feet) thick. This range allows percolating water an opportunity to react chemically with its environment before reaching ground water. Percolating water has four options in passing through the unsaturated zone. It can move virtually unchanged, can show a net gain of solute, show a net loss of solute, or keep the same total ionic concentration with a net exchange of ions. Since few soils or sediments are chemically inert, changes in transported solute are to be expected.

Chemical activity in the unsaturated zone is primarily located at the surfaces of clay minerals and hydrous oxide coating. Limited microbial activity may take place either from the indigenous population or that transported from solid waste.

Cations will be removed from solution until either the cation exchange capacity is reached, or the limit of displacement reactions is reached. The limit of cation exchange capacity (CEC) can range from nearly zero to probably not more than 60 milliequivalents per 100 grams of soil. Solution concentrations, pH, and percolation rate quantitatively affect the reactions. It should be noted that adsorption is not a permanent fixation. Cations may be desorbed with changes in solution composition, pH, or oxidation-reduction (redox) potential.

In soils and sediments underlying landfills, the cation exchange capacity will immobilize a certain amount of the leached cations. When the capapcity has been reached, further percolaton of cations will not be affected. Adding cover material of a heavier texture than sand should help attenuation by providing exchange capability between solid waste cells.

Divalent and trivalent cations include most of the heavy metals. These are held more strongly than sodium, potassium, or ammonium on the cation exchange complex. Divalent and trivalent cations will displace monovalent cations which are adsorbed.

Heavy metals are prone to adsorption on hydrous oxide coatings in the soil. The hydrous oxides are frequently cited as so limiting metal solubility that agricultural deficiencies of copper, zinc, and cobalt occur.[2] Attenuation of heavy metals present in leachate is desirable. In locations virtually free of clay minerals, these coatings may be present on sand grains giving the sandy formation some ability to attenuate metallic ions.

Adsorption is only one mechanism for removing dissolved ions from solution. Changes in the geochemical environment can also affect solution equilibria. A transition from reducing conditions in the landfill to oxidizing condition in the unsaturated zone can reduce the concentration of some redoxsensitive species in solution and change the chemical form of others. For example, iron and manganese will oxidize and precipitate from solution.

If porosity will allow bacterial movement, biochemical reactions involving leachate constituents can proceed. Sulfide and ammonium can be oxidized to sulfate and nitrate. Dissolved organic matter measured in terms of BOD and COD can be reduced through microbial decompostion. Some nutrient elements in the course of these reactions will be incorporated in bacterial cells and thereby be removed from solution until the bacterial cells die off. Conversion of ammonium to nitrate changes nitrogen from a form subject to attenuation to a form which is not. Sulfide to sulfate oxidation is not expected to be as significant. Sulfide can form insoluble precipitates with many of the heavy metals. For this reason, it may not be present in more than trace amounts in leachate.

Microorganisms may also attack the organic ligands associated with chelated and complexed metals. Decomposition or absorption by microorganisms would remove the metals from leachate.

Phosphate reacts with a variety of soil components forming insoluble products. Calcium and phosphate react in solution to form hydroxypapite, the least soluble phosphate compound known. Iron, alumninum, and manganese can also form virtually insoluble precipitates with phosphate. These reactions lead to a strong attenuation of phosphate when these metal ions are present in the unsaturated zone.[3]

Carbonate also reacts with calcium, magnesium, and some heavy metals forming relatively insoluble compounds. Calcareous deposits in the unsaturated zone can be valuable in attenuating phosphate and heavy metals from leachate. Because carbonate neutralizes acids, BOD and COD as expressed in organic acid concentration may also be reduced. Carbonate induced alkalinity may change solubilities of heavy metal chelates and lead to a deposition of heavy metals.

The unsaturated zone has a reciprocal influence on the leachate which percolates into it. Water of low oxidation potential first infiltrating into the unsaturated zone of high oxidation potential will become more oxidized while simultaneously reducing substances in the unsaturated zone. A continued percolation of reduced water may convert what had been an oxidized system into a reduced one. Or, the percolate may become oxidized if that capacity in the unsaturated zone is greater. The degree of influence of reduced leachate on the oxidized unsaturated zone and vice versa depends upon the reserves of material capable of oxidizing or reducing in the unsaturated zone and leachate. The greater the distance leachate travels between solid waste and ground water, the better the chance that the entire path through the unsaturated zone will not become reduced. Raising the oxidation potential of leachate will tend to attenuate some components in solution at the point of exit from the solid waste.

5.2.3 Aquifer Zone.

Concepts useful for describing surface-water pollution are generally not valid for ground water. Ground-water movement is described by Darcy's Law which states that velocity is directly proportional to the permeability of the aquifer and the hydraulic gradient and inversely proportional to the porosity. Ground-water flow velocities vary over a wide range, 1.5 meters/year to 1.5 meters/day, typically 5 feet/year to 5 feet/day). Highly permeable glacial outwash deposits, fractured basalts and granites, and cavernous limestone aquifers allow much greater velocities.

The relatively slow velocity of ground water results in laminar flow which exhibits different characteristics of mixing than does turbulent flow usually associated with surface streams. A water of different chemcial composition from ground water injected or percolated into ground water tends to maintain its integrity and is not diluted with the entire body of ground water. Instead, it moves with the ground-water flow as a plume undergoing minimal mixing.

The plume shape is determined by the physical characteristics of the aquifer. Porous media give somewhat different shaped plumes from fractured rock or cavernous limestone. Chapter 2 illustrates the paths of ground-water movement in various hydrologic regimes.

Differential attenuation is defined as a reduction in concentration of a dissolved constituent, with distance along the direction of water flow which is disproportional to changes in concentration of other constituents. Differential attenuation may result from chemical or biochemical reations which remove the constituent from solution. Apparent attenuation occurs from dilution by mixing with water of lower constituent concentration.

Dilution may take place in ground water in two ways. One is hydrodynamic dispersion, and the other is molecular diffusion. Microscopic dispersion describes mixing caused by the tortuous flow of water around individual grains and through pores of various sizes in a porous aquifer. Macroscopic dispersion describes mixing as water flows in and around heterogeneous geologic formations. Molecular diffusion is the diffusion of solute across a concentration gradient from stronger to weaker concentration. It is seldom possible to measure in the field. On the other hand, there are mathematical equations which describe dispersion. By measuring enough physical and chemical parameters at a site over a sufficient length of time, an approximaate calculation can be made for dispersion.

Chemical interactions provide the greatest amount of differential attenuation in the aquifer zone. Hydrous oxides of iron, aluminum, and manganese or clay minerals present in aquifers attenuate cations in the same way that they do in soils or in the unsaturated zone. Since hydrous oxide and clay colloids are in constant contact with water in the aquifer, it can be assumed that the exchange sites are saturated and essentailly in equilibrium with the ambient ground water. When contacting these colloids, leachate-enriched ground water will initiate cation exchange resulting in desorption of cations which are less strongly held than those replacing them. In this way, hydrogen, sodium, calcium and magnesium may be released into the aqueous phase by exchange with heavy metals and other cations in leachate. High hardness values associated with leachate plumes may be due, in part, to this ion exchange phenomenon.

Chemical precipitation in the aquifer is possible if the natural ground-water composition includes ions which form insoluble compounds with constituents in leachate. An example would be formation of hydroxyapatite with leachate phosphate and calcium in ground water.

Changes in redox potential, buffering reactions, or changes in lithology may produce other attenuation reactions.

The third means of attenuation in aquifers is that termed decay. Oxidation of organic compounds reduced them to carbon dioxide and water. Microorganisms carried into the aquifer zone are deprived of a good nutrient supply and are subjected to a cooler temperature than in the solid waste zone. This results in a lowering of biochemical activity, frequently to the point of cessation. The inactivation coupled with natural die off tends to reduce bacterial numbers rather rapidly.

There are two additional complications in the interpretation of ground-water quality in leachate plumes. One is the variation in leachate concentration with time, and the other is the discontinuous recharge of leachate which occurs in most geographical regions.

Leachate production begins as soon as deposited solid waste attains field capacity; this can occur locally in channelized patterns or integrally. The lag time depends upon local climatic conditions and rate of solid waste deposition. In an active landfill, older organic matter is stabilizing while simultaneously new organic matter is beginning to ferment and produce stronger leachate. The net effect is an increasing leachate concentration from a given area and/or increasing areal contamination as long as the landfill is active.

Leachate produced at the initiation of percolation through the landfill is less concentrated than that produced after several years' solid waste accumulation. This leachate will be found at the distal end of the plume of leachate-contaminated ground water. The closer the sampling point to the landfill, a more concentrated contaminated ground-water would be expected. An increasingly concentrated leachate source in addition to the factors of dilution and attenuation must be considered in interpreting the results of sampling the plume. If the variation in source strength is ignored, an erroneously high value for attenuation or dilution may be given.

The intermittent recharge occurring from most landfills also complicates interpretation of leachate-plume configuration. During summer months when evaporation frequently exceeds rainfall, little or no leachate may be produced. Repeated infiltration of precipitation from summer showers into solid wastes should produce a zone of leached and deposited contaminants. When evaporative losses are reduced in the fall, percolating water will then re-dissolve these materials. They will then travel as a relatively highly concentrated slug to the ground-water reservoir. Ground water, however, moves under the landfill at a relatively steady rate. Thus, there will be temporal variations in the volume and strength of leachate reaching ground water. These variations will show in the leachate plume as variations in total solute concentration. A sample taken from the plume at any given time may represent a "high" or "low" in the intermittent recharge pattern. One way to visualize this phenomenon would be to observe the response of a conductivity probe in a monitoring well over a period of time. As leachate-enriched ground water move past the well, conductivity will vary with changes in dissolved solids concentration. The variations may be noticeable only in time spans of weeks to months. Since concentrations may vary from factors other than aquifer characteristics, calculations of values for dispersivitiy in dilution may not be accurate.

A generalized summary of the susceptibility of leachate to differential attenuation is provided in Table 12. The mechanism of attenuation which affects each constituent is listed for the zones through which leachate may pass. When data are summarized in this fashion, only the principal mechanisms can be cited. For example, "no attenuation" is listed for all of the constituents in the solid waste zone. However, this is not always valid. Quantification is impossible, and there is a net release of most of the constituents. Sulfate, nitrate, and ammonium are given biochemical conversion alternatives. These ions are subject to oxidation and reduction reactions which may convert or eliminate them. Heavy metals are also prone to one or more of the attenuation mechanisms and may not be universally present in leachate. Biochemical reactions for the aquifer zone are not listed as biological activity is inhibited. In places, biological activity may be significant in the aquifer; but the amount and type cannot be predicted.

5.2.4 Measurement of Attenuation.

From the previous discussion, it is evident that attenuation describes two phenomena associated with the transportation of solute. One is dilution resulting from dispersion and diffusion; the other is dilution resulting from chemical or biochemical removal of solute from ground water. The former type of dilution is referred to as apparent attenuation since no active chemical processes are operating to reduce the concentration of dissolved constituents.

Since the prediction of the future conditions depends upon the extent of active attenuation, it is important to distinguish between apparent and active attenuation in the field. To accomplish this, several samples of leachate-enriched ground water must be collected along the leachate plume's path of travel. Chemical constituents measured in these samples are then chosen on the basis of their susceptibility to attentuation. Relative changes in concentration with distance from the source are noted. Providing background levels are not influenced by other sources of chloride contamination (e.g. road salting, brine dumping, etc.), chloride is the best constituent to measure as an indicator of dilution. Since it carries a negative charge and does not form precipitates with the common cations in

TABLE 12

SUSCEPTIBILITY OF LEACHATE CONSTITUENTS TO DIFFERENTIAL ATTENUATION

Attenuated Constituent	Solid Waste Zone	Unsaturated Zone	Aquifer
Chloride	0	0	0
Sulfate	0-B	0-B	0
Sulfide	C	C-B	C
Phosphate	0	A-C	A-C
Nitrate	0-B	0	0
Ammonium	0-B	A-B	A
Sodium	0-B	0	0
Potassium	0	A	A
Calcium	0	A	A
Magnesium	0	A	A
Heavy Metal Anions (Cr, V, Se, B, As)	0-B	0-B	0
Heavy Metal Cations (Pb, Cu, Ni, Zn, Cd, Fe, Mn, Hg)	0-A-C	A-C	A-C
Organic Nitrogen	0	B	0
COD	0	B	0
BOD	0	B	0
Volatile Acids	0	B	0
Phenols	0	A-B	0-A
MBAS	0	A-B	0-A

0= No Attenuation A= Adsorption B= Biochemical Degradation On Conversion
C= Chemical Precipitation

water, chloride is unaffected by ambient conditions. Reductions in chloride concentration can then be attributed to dispersion and diffusion. If ground-water equations were used in an attempt to calculate dispersion coefficients for leachate-enriched ground water, chloride concentration data would be a good first choice for use in the calculations. Nitrate reacts in virtually the same way, but nitrate is less frequently present in leachate in comparable concentrations.

Concentrations of other constituents sampled simultaneously with chloride should represent equal dilution. If they are observed in lower than expected concentrations, this indicates that active attenuation has taken place. Conversely, if their concentrations are greater than those calculated on the basis of chloride, desorption from ion exchange sites or contributions from other sources may account for the nontheoretical results.

An example which is calculated from data obtained in a landfill study is presented below.[4] The cations calcium, sodium plus potassium, ammonium, and iron in leachate-enriched ground water are plotted in percentage of original concentration vs. distance from the landfill (Figure 54). If all of the cations were diluted equally, they would plot on the same curve. Reference points for chloride are included to facilitate a comparison of the theoretical dilution-only curve with the actual cation concentration curves.

The plume of leachate-enriched ground water represented by Figure 54 is produced by a landfill that has been active for 28 years. The leachate plume can be traced about 3231 meter (10,600 feet) downgradient from the landfill and vertically throughout the thickness of the aquifer (about 24 meters—80 feet) with the most concentrated contamination near the bottom.

Calcium remains above the chloride curve throughout the length of the plume in agreement with other reports indicating that calcium is desorbed from clays as a result of cation exchange with leachate components.[5] In this specific situation, there may also be a contribution of calcium from septic tank effluent.

Ammonium remains above the chloride curve for about half the length of the plume. It also may be desorbed and is a component of septic tank effluent. The loss of ammonium at more distant points of the plume may be due, in part, to more aerobic ground-water conditions that will permit nitrification of ammonium.

Sodium and potassium generally plot below the chloride points, and iron is even more attenuated. Probably the iron is removed largely through solubility changes resulting from increases in Eh as distance from the landfill increases.

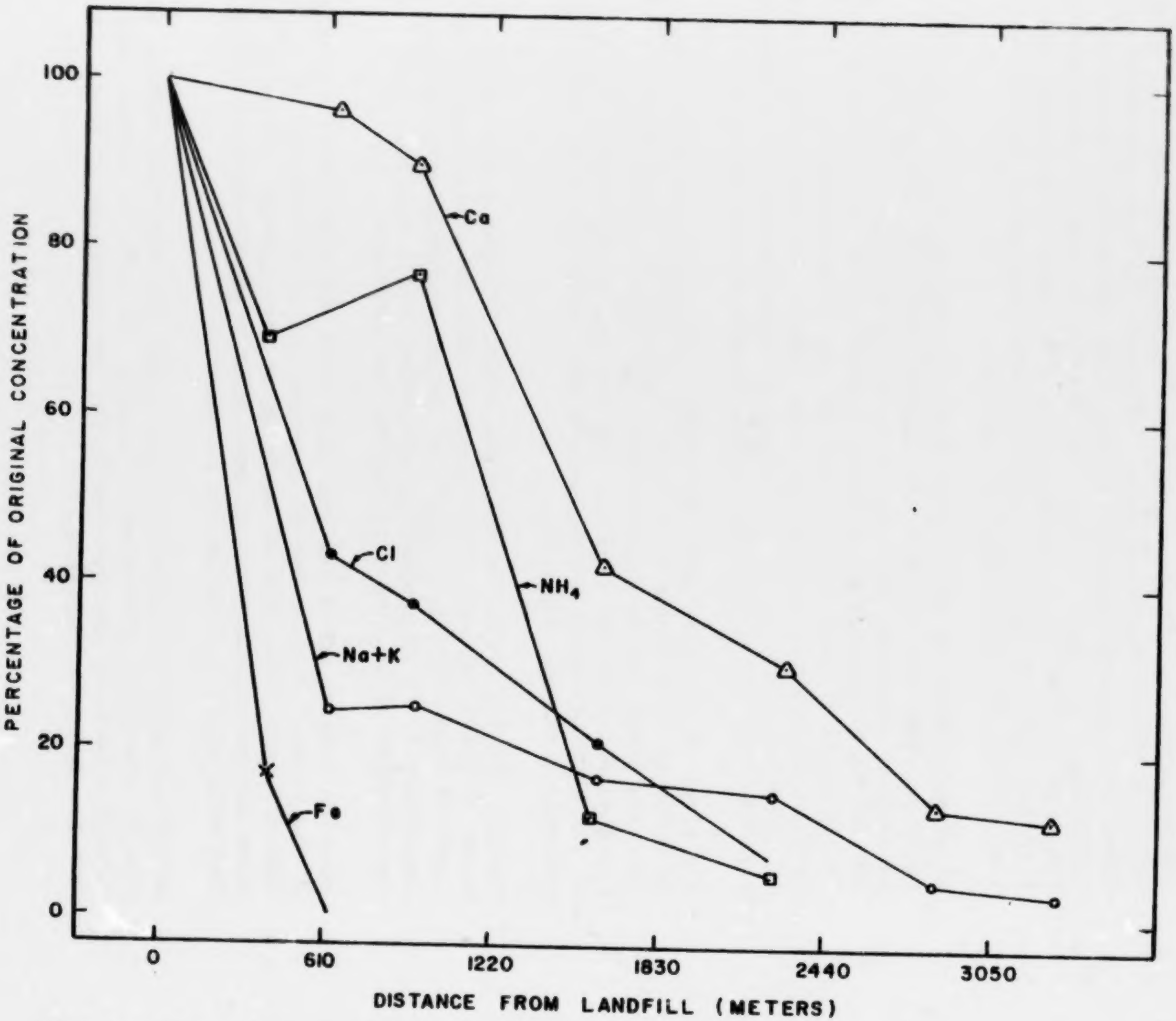


FIGURE 51. CATION CONCENTRATIONS AS A PERCENTAGE OF ORIGINAL CONCENTRATION WITH DISTANCE FROM A LANDFILL

The geohydrologic environment is characterized by soft, rather acid native ground water in a highly silicaceous, unconsolidated sand and gravel aquifer. Uncontaminated ground water contains iron in concentrations which frequently exceed the recommended drinking water limit of 0.05 mg/l. No significant amounts of clay or silt are present in the path of the leachate plume. Sand grains are coated with iron oxide which probably exhibits a small amount of cation exchange capacity. Septic tanks used in the area and intermittent recharge of leachate as influenced by the climate complicate the interpretation of chemical data along purely theoretical principles.

5.3 LEACHATE QUANTITY

Estimating the leachate quantity being generated at a land disposal site is important in the environmental assessment of the site and the relative need for monitoring the site. Knowledge of leachate quantity and generation rates will also be useful in determining which monitoring scheme and pollution abatement programs should be most effective.

The water-balance or water-budget method has been presented in the literature as a useful tool in estimating leachate quantities and generation rates at land disposal sites.

In two U. S. Environmental Protection Agency reports,[1,6] excellent summaries of the water balance method and its application to land disposal sites are presented. The reports discuss the theories and principal factors involved in leachate generation including useful information and data on the following:

- The influence of slope, surface condition, and soil type of the quantity of runoff and the potential for leachage production;
- The dependency of infiltration on the storm frequency, duration, intensity, and soil moisture conditions;
- The influence of vegetation on evapotranspiration and infiltration;
- The relationship of soil permeabilities to infiltration rates and volumes;
- The moisture retention capabilities of various types of solids as well as compacted municipal solid waste.

Caution must be exercised in applying the water-balance method to land disposal sites. Review of the above-referenced information clearly shows the extreme sensitivity of leachate quantity estimates to the many variables used in the water-balance calculations. For example, slight changes in runoff coefficients, evapotranspiration, or moisture retention figures can result in a significant change in the leachate quantity estimate. In addition, unless extensive on-site measurements are performed, the many parameters in the water-balance calculations are purely theoretical estimates. Therefore, this manual presents the water balance method as a useful tool for planning, design, and assessment purposes. The leachate quantity estimates generated should be viewed with this qualification in mind.

5.3.1 Water Balance Simplified

The water-balance or water-budget method is the measurement of the continuity of flow of water for any given time interval and can be applied to any drainage basin.[7] Here, the drainage basin being considered is a land disposal site together with the land immediately surrounding it. The calculation of the water balance for a landfill requires the measurement of numerous physical parameters and can be a relatively difficult and expensive task. For most landfill investigation and monitoring work, however, a reasonable approximation of the magnitude of the various water-balance components will be sufficient. Methods of estimating each of these components, using as much available information and as few field measurements as possible, are given below.

The seven principal water-balance components of a hypothetical landfill are shown in Figure 55. These are: precipitation and irrigation, surface runoff onto the landfill, surface runoff from the landfill, evapotranspiration, underflow (in and out), and infiltration.

Precipitation and Irrigation. In most cases, precipitation and irrigation will be the principal source of water into the landfill. This may not be true for landfills placed in the water table; landfills not diverting runoff from adjacent property; or landfills accepting large quantities of liquid wastes. Significant variations in precipitation may occur in certain localized areas, especially in mountainous regions requiring cautious use of generalized rainfall maps. Significant variations may also occur with time, an abnormally wet year (for example); such abnormalities cannot be shown on a general map. For these reasons it is advisable to seek precipitation data specific to the landfill site and for a number of years preceeding the investigation. Historical precipitation records for weather stations can be obtained from the U. S. Department of Commerce, National Oceanic and Atmospheric Administration, Environmental Data Service, Asheville, North Carolina. The locations

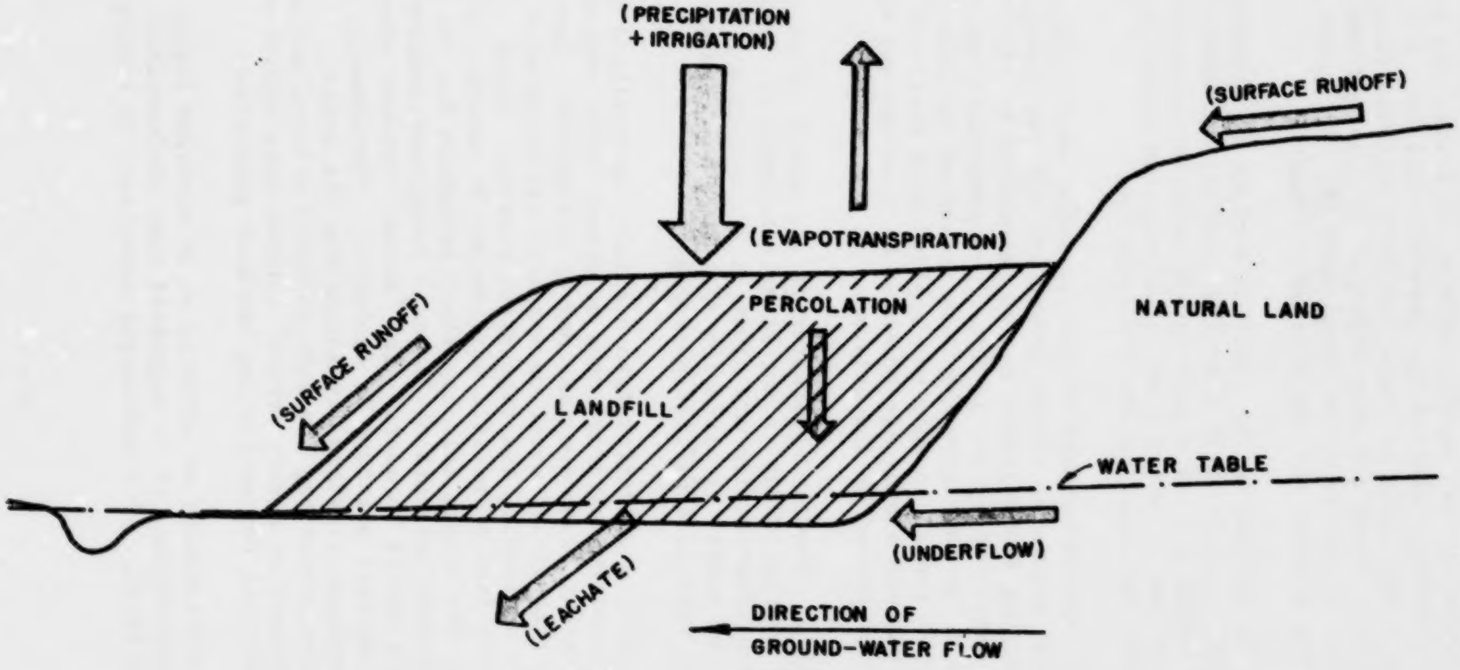

(PRECIPITATION
+ IRRIGATION)
(SURFACE RUNOFF)
(EVAPOTRANSPIRATION)
PERCOLATION
NATURAL LAND
(SURFACE RUNOFF)
LANDFILL
WATER TABLE
(UNDERFLOW)
(LEACHATE)
DIRECTION OF
GROUND-WATER FLOW

FIGURE 55. LANDFILL WATER BALANCE SIMPLIFIED

of weather stations for which data are available are shown on maps obtainable from the above address. Interpolation of the data from two or more stations can be made to more closely approximate the precipitation at the landfill site. For extended investigations or monitoring programs, it may be desirable to determine the precise volume of precipitation reaching the landfill surface. For this purpose, a rain guage would be installed in a suitable location on or near the landfill. There are many types of rain gauges available, and the selection of one would be based upon the particular conditions of the monitoring program and the available budget.

Irrigation may be applied to the landfill surface to maintain a desired vegetation growth, particularly when the landfill is completed and its surface is being used as a golf course or other recreational facility. The volume of water used for irrigation should be measured with a flow meter and added to the precipitation.

Surface Runoff. The percentage of precipitation which flows onto the landfill from adjoining higher ground and off the landfill surface to adjoining lower ground can be calculated by the rational runoff formula described by Ven Te Chow.[7] A reasonable estimation of runoff can also be made from the data presented in Table 13, where the rational runoff formula was applied to a series of typical situations. Areas and slopes are measured by a survey, and surface conditions are determined by inspection.

Evapotranspiration. Evapotranspiration is the sum of water loss by evaporation and transpiration (plant water consumption). Methods of calculating evapotranspiration are given in the hydrologic literature.[7] However, the large number of variables that must be measured to perform the calculations make it a difficult process.

Estimation of evapotranspiration from available generalized data, such as "potential" evapotranspiration maps or annual water consumption figures for different plant species, may be misleading. This approach cannot account for numerous specific variables such as soil type, soil water available, and vegetation density. Since evapotranspiration from a landfill surface may be anywhere from insignificant to the single most important mechanism for the removal of water, an accurate estimate of "actual" evapotranspiration from the specific site should be made. (The "actual" becomes less than the "potential" as soil moisture is depleted.) Thornthwaite and Mather[9] have developed a method for estimating the actual evapotranspiration accounting for soil moisture effects and an Environmental Protection Agency report[6] applies this method to hypothetical landfills for estimating leachate generation.

Because of the difficulties in arriving at an accurate figure for actual evapotranspiration, it is suggested that professtional assistance be obtained. If a hydrologic consultant is retained

TABLE 13

PERCENTAGES OF SURFACE RUNOFF FOR A 2.5 cm RAINFALL (8)

Surface Condition	Percent Slope	Percent Surface Runoff Sandy Loam	 Clay or Silt Loam	 Clay
Pasture or meadow cover crop				
Flat	0 - 5	10	30	40
Rolling	5 - 10	16	36	55
Hilly	10 - 30	22	42	60
No vegetation-not compacted				
Flat	0 - 5	30	50	60
Rolling	5 - 10	40	60	70
Hilly	10 - 30	52	72	82

for the landfill study, he will be able to estimate actual evapotranspiration for the specific case involved. If such a consultant is not used, information on evapotranspiration rates for an area will often be available from a local agricultural research station, a nearby U.S. Geological Survey field office, or possibly the agriculture department of a nearby university.

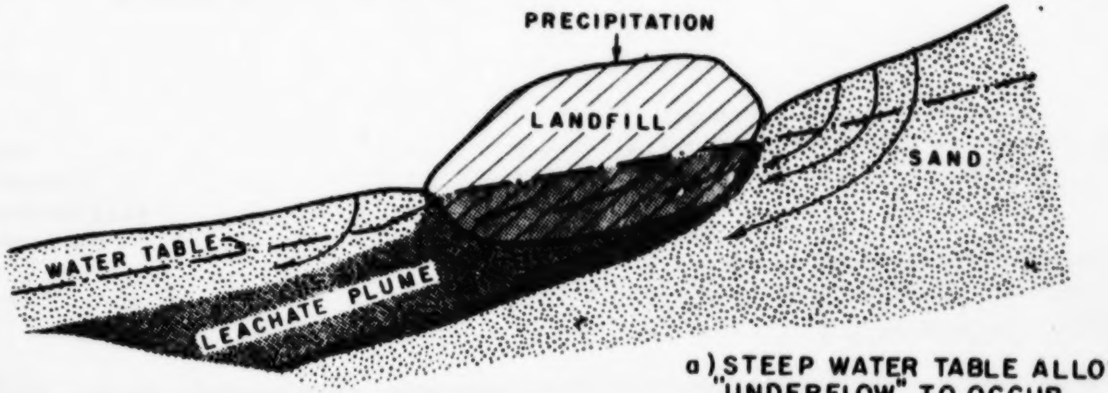
Underflow Underflow is defined here as the rate of ground-water flow from adjoining areas directly into the landfill. This condition will occur only if the base of the landfill is below the water table. A second necessary condition, however, is that the landfill adjoins or is situated near an area of elevation substantially higher than its base--that is, there is a significant water-table gradient beneath the landfill. If the landfill is situated on level ground and substantial percolation of water through the landfill is occuring, leachate being generated by the percolation will move away from the landfill and underflow, as defined above, will not occur (Figure 56).

Precise measurement of underflow, if it is occuring is not feasible. A determination of the occurrence of underflow, and a reasonable approximation of its rate can be made, however, by means of a relatively straightforward hydrologic investigation. Figure 57 presents a schematic diagram and method for estimating the rate of underflow. This process requires the drilling and testing of at least two wells, which could also be used as monitoring wells as part of the on-going monitoring program.

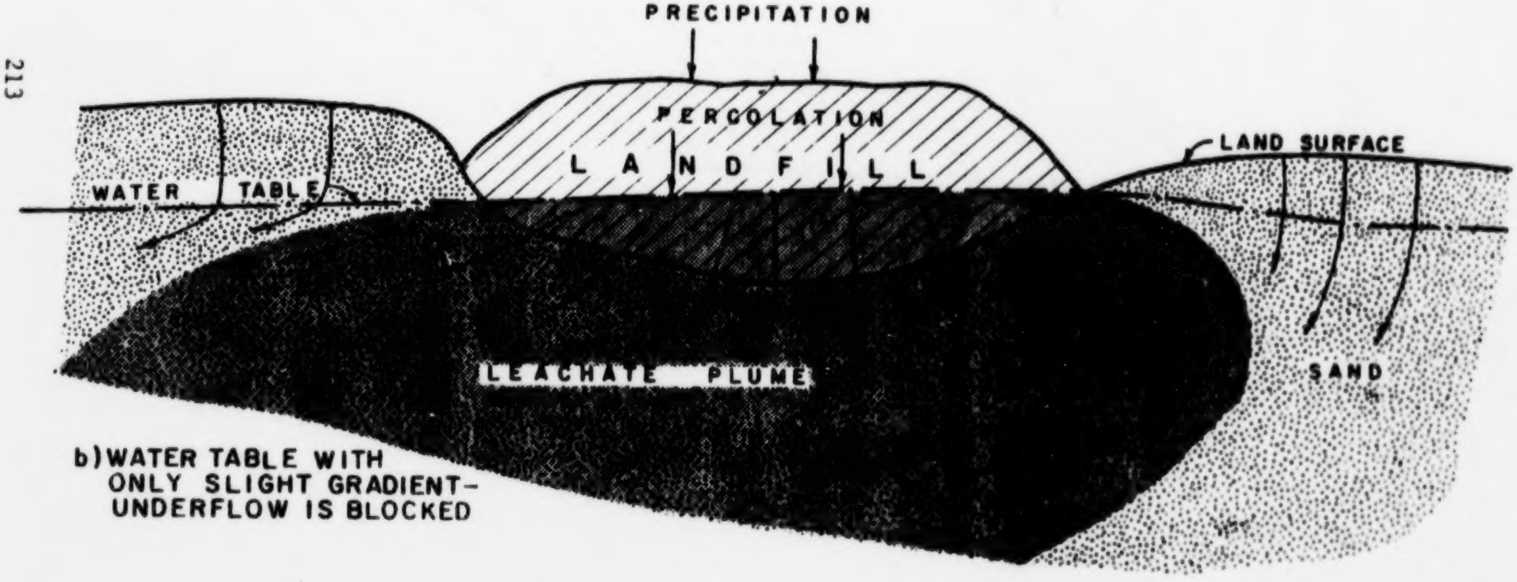
Percolation. Infiltration is generally defined as that portion of surface water which penetrates the land surface. Some of this water may then be taken up by plants and some may be directly evaporated from the first few centimeters (inches) of soil. That portion of water which migrates below the root zone and into the solid waste and thus contributes to the volume of leachate generated is the percolate. However, solid waste placed in a landfill is capable of absorbing a certain volume of water and holding it against the force of gravity. The volume of water so absorbed by the solid waste is termed its field capacity. Generally, municipal solid waste field capacity is about 25.4 millimeters (1 inch) of water for each .3 meters (1 foot) of solid waste. Leachate will not be generated in any significant volume until almost all of the solid waste has reached field capacity. Channeling may occur resulting in some leachate generating prior to reaching field capacity.

Percolation can be calculated as follows:

Percolation = precipitation + runoff onto landfill - runoff from landfill - actual evapotranspiration.

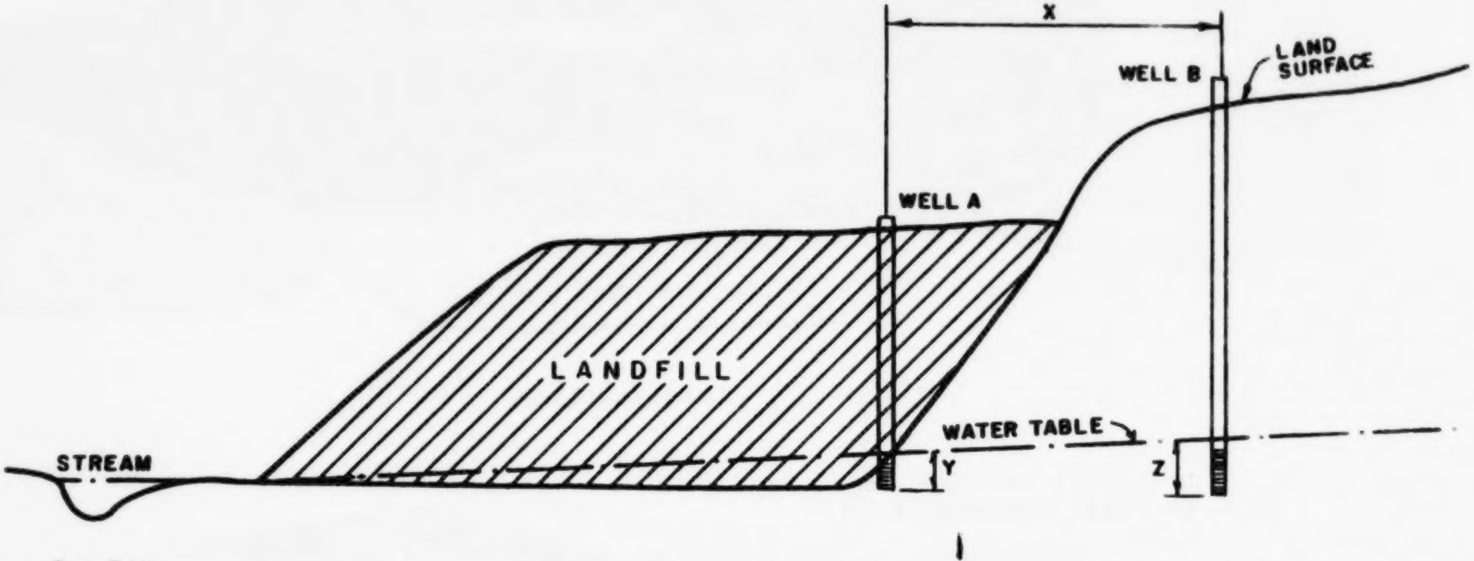

PRECIPITATION
LANDFILL
SAND
WATER TABLE
LEACHATE PLUME

a) STEEP WATER TABLE ALLOWS "UNDERFLOW" TO OCCUR


PRECIPITATION
PERCOLATION
LANDFILL
LAND SURFACE
WATER TABLE
LEACHATE PLUME
SAND

b) WATER TABLE WITH ONLY SLIGHT GRADIENT- UNDERFLOW IS BLOCKED

FIGURE 56. CONDITIONS AFFECTING UNDERFLOW.


X
WELL B
LAND SURFACE
WELL A
LANDFILL
WATER TABLE
STREAM
Y
Z

Q = PIA
Where:
Q = Underflow
P = Permeability
I = Gradient
A = Area

P is estimated by pumping tests or injection tests in Wells A and B, or by examination of geologic samples removed during drilling

I = [z (meters) - y(meters)] ÷ x(meters)
A = (y) x (horizontal length of contact between the landfill and high ground)

FIGURE 57. ESTIMATING RATE OF LANDFILL UNDERFLOW

Direct measurement of percolation is possible using a sub-surface water trap such as the one shown on Figure 58.

Leachate Generation. Having estimated all of the pertinent parameters, leachate generation can be simply estimated with the following water balance formula:

Leachate generation = percolation + Underflow

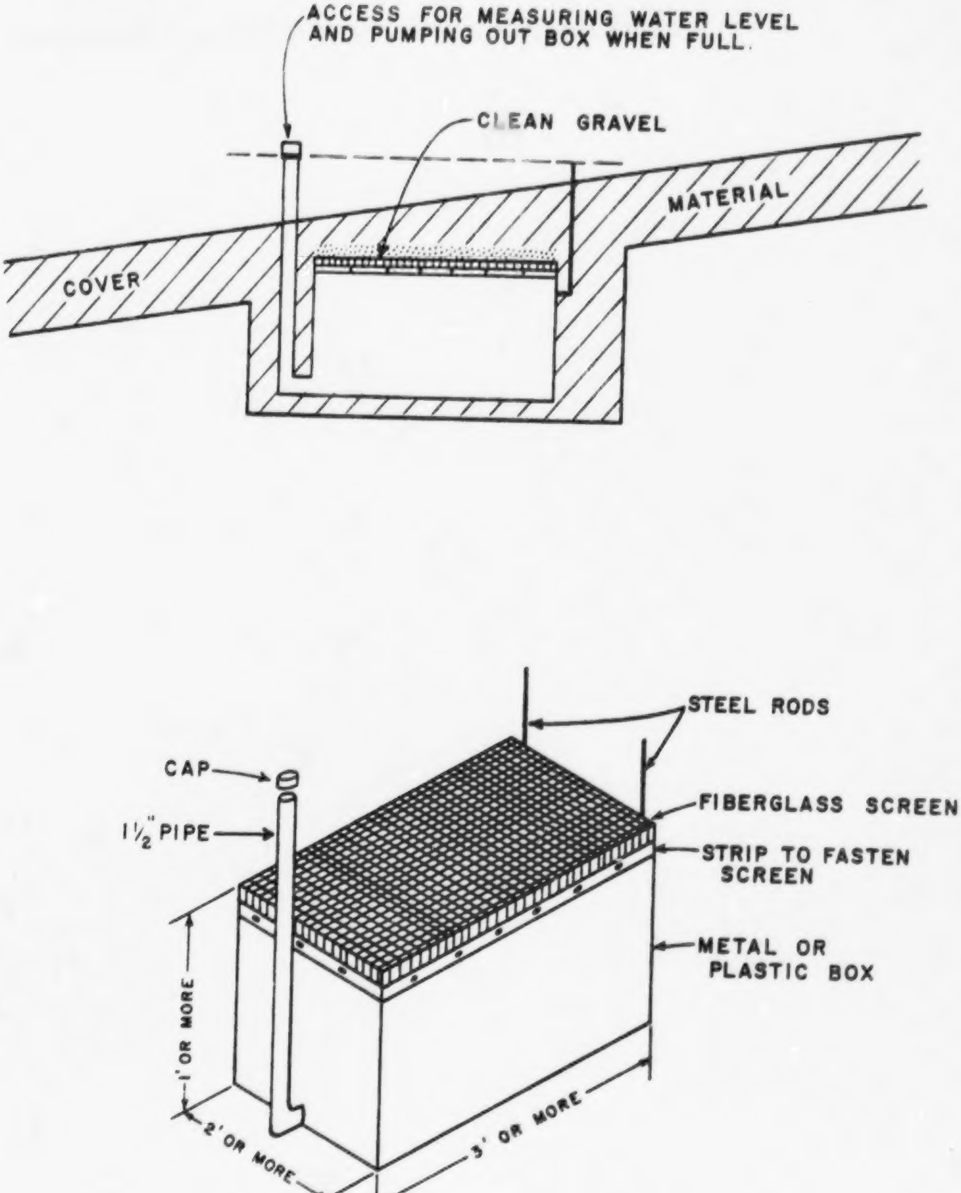

ACCESS FOR MEASURING WATER LEVEL AND PUMPING OUT BOX WHEN FULL.
CLEAN GRAVEL
MATERIAL
COVER
STEEL RODS
CAP
FIBERGLASS SCREEN
1½" PIPE
STRIP TO FASTEN SCREEN
METAL OR PLASTIC BOX
1' OR MORE
2' OR MORE
3' OR MORE

FIGURE 58. DEVICE FOR DIRECT MEASUREMENT OF LANDFILL INFILTRATION

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6.0 SAMPLE WITHDRAWAL, STORAGE, AND PRESERVATION

6.1 INTRODUCTION

The sampling of ground and surface waters associated with the monitoring of land disposal sites is a critically important operation. The analytical results obtained from the samples and the subsequent decisions based upon the analytical data are vitally dependent upon the validity of the samples obtained.

Every effort must be made to assure that the sample is representative of the particular body of water being sampled. A detailed sampling plan, acceptable to all interested parties, should be developed prior to any sampling operations.

The physical, chemical, and bacteriological integrity of the sample must be maintained from the time of sampling to the time of testing in order to keep any changes at a minimum. The time between sampling and testing should be kept at the absolute minimum.

6.2 SAMPLE COLLECTION

6.2.1 Sample Withdrawal Methods

Sample withdrawal methods include the use of pumps, compressed air, bailings, and samplers. The primary consideration is to obtain a representative sample of the ground-water body by guarding against mixing the sample with stagnant (standing) water in the well casing. In a non-pumping well, there will be little or no vertical mixing of the water; and stratification will occur. The well water in the screened section will mix with the ground water due to normal flow patterns, but the well water above the screened section will remain isolated and become stagnant. Persons sampling should realize that stagnant water may contain foreign material inadvertently or deliberately introduced from the surface, resulting in unrepresentative data and misleading interpretation of the same.

To safeguard against collecting non-representative stagnant water in a sample, the following guidelines and techniques should be adhered to during sample withdrawal:

> As a general rule, all monitoring wells should be pumped or bailed prior to withdrawing a sample. Evacuation of a minimum of one volume of water in

- the well casing and preferably three to five volumes is recommended for a representative sample. In a high-yielding ground-water formation and where there is no stagnant water in the well above the screened section, evacuation prior to sample withdrawal is not as critical. However, in all cases where the monitoring data is to be used for enforcement actions, evacuation is recommended.

- For wells that can be pumped or bailed to dryness with the sampling equipment being used, the well should be evacuated and allowed to recover prior to sample withdrawal. If the recovery rate is fairly rapid and time allows, evacuation of more than one volume of water is preferred.

- For high-yielding monitoring wells which cannot be evacuated to dryness, bailing without pre-pumping the well is not recommended; there is no absolute safeguard against contaminating the sample with stagnant water. The following procedures should be used:

  a. The inlet line of the sampling pump should be placed just below the surface of the well water and three to five volumes of water pumped at a rate equal to the well's recovery rate. This provides reasonable assurance that all stagnant water has been evacuated and that the sample will be representative of the ground-water body at that time. The sample can then be collected directly from the pump discharge line.

  b. The inlet line of the sampling pump (or the submersible pump itself) should be placed near the bottom of the screen section, pumped approximately one well volume of water at the well's recovery rate, and the sample collected directly from the discharge line.

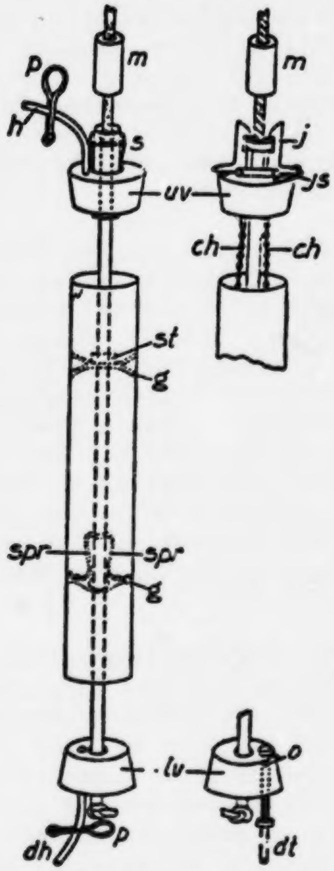
A non-representative sample can also result from excessive pre-pumping of the monitoring well. Stratification of the leachate concentrations in the ground-water formation may occur, and excessive pumping can dilute or increase the contaminant concentrations from what is representative of the sampling point of interest.

In light of this discussion, bailing by hand is not a recommended well-sampling method unless adequate precautions are taken. Bailing is accomplished in small diameter wells by lowering and raising a weighted bottle or capped length of pipe on a length of rope. Rarely can a sufficient quantity of water be removed to adequately eliminate stagnant water from the sample unless many time-consuming trips in and out with the bailer are made. Persons sampling a well often will use the first bailer full of water as the sample because of the ease with which the sample can be colledted. The reliability of this sample is nul, and this fact must be impressed upon the sample collector. Of course, in situations where the well can be bailed dry or there is only about a meter of water in the bottom of a well representative samples can be obtained with a bailer as the casing can readily be emptied.

A Kemmerer water bottle sampler, as shown on Figure 59, is a commonly used bailer. In transferring the sample from the sampler to the sample bottle, contact with air and agitation of the sample should be minimized; slow and careful transfer, placing the tip of the sampler's exit tube to the side of the sample bottle, is recommended. To minimize cross-contamination, the bailers should be thoroughly flushed out with tap water and the first sample from the next well to be sampled prior to collecting the sample for analysis.

Where the water table is within suction lift, small-diameter wells can be sampled with peristaltic, centrifugal, or pitcher pumps. Peristaltic pumps have rather low pumping capacities. The sample is conducted through inert silicone rubber tubing, thereby reducing the possibility of sample contamination by constituents from the sampling apparatus. For this reason, the procedure is attractive. Small, highly portable centrifugal pumps are available with pumping rates from .3 to 2.5 l/s (5 to 40 gpm). Removing stagnant water and flushing the discharge set-up clean will pose little difficulty, allowing collection of representative samples. If concentrations of less than one ppm are being investigated, extra care must be taken in sampling; in the extreme ppb range, the peristaltic pump would have to be used.

Pitcher pumps can be easily carried to a site, screwed onto a well, and used to pump a sample. No power source is required other than the investigator, and equipment costs are low. A typical vacuum sampling technique would involve the use of a vacuum or suction pump and a portable generator as a power source. Vacuum can also be supplied from an automobile or truck engine manifold or from a hand pump, replacing the vacuum pump. In Orange County Florida[1], such a vacuum pump was used to collect ground-water samples under anaerobic conditions. At that location, a 1.27-cm (1/2-inch) tube is permanently installed in the monitoring well thereby eliminating the possibility of cross-contamination between wells.



ch—chain which anchors upper valve to upper interior guide.
dh—rubber drain tube.
dt—brass drain tube.
g—interior guide fastened to inner surface of sampler.
h—rubber tube.
j—jaw of release.
js—jaw spring.
lv—lower valve.
m—messenger.
o—opening interior of drain tube.
p—pinch cock.
s—upper release spring operating on horizontal pin, one end of which fits into groove on central rod.
spr—spring fastened to lower internal guide and operating in groove on central rod to provide lower release.
st—stop on central rod.
uv—upper valve.
Left—View of complete sampler with valves open.
Top right—Another type of construction of upper valve and tripping device.
Bottom right—Another type of construction of lower valve and drain tube.

FIGURE 59. STRUCTURAL FEATURES OF MODIFIED KEMMERER SAMPLER

(P.S. Welch, Limnological Methods. p. 200. Figure 59.)

As an alternative to these pumps, an inexpensive bailer pump can be constructed from readily available materials. This pump consists of a length of garden hose with a foot valve at its bottom end and fittings at its top end that allows a vacuum to be applied to the base. A water sample is collected by moving the hose up and down and activating the foot valve, causing the partial vacuum to assist in bringing water to the surface. Vacuum is obtained from an automobile engine. It should be remembered, however, that this sampler should be used where the well contains only a small volume of water in order that clearing stagnant water from the casing does not become an inordinately time-consuming process.

An inexpensive air lift sampler can be constructed from polyethylene or any reasonably flexible tubing, as shown in Figure 60. Since the tubing is flexible, it can be readily coiled and moved conveniently from well to well. Primary limitations on the sampler are the amount of air pressure that can be safely applied to the tubing and finding a suitable source of compressed air. A high-pressure hand pump would serve nicely for a shallow water table, but a small air compressor may be required for lift greater than 9 meters (30 feet). The advantage of this sampler is that the apparatus can be designed for permanent installation in the monitoring well. The smaller the tubing used, the less that can be pumped; this can be compensated for by pumping longer to obtain the minimum sample volume required for analysis.

With a pressure-type sampling method, the sample is obtained by connecting the sample bottle directly to the 1.27cm (½-inch) water discharge outlet. To prevent contact with oxygen, the sample bottle could be flushed out with an inert gas prior to collecting a sample. The built-in feature of this method used to both pre-pump and sample the well can effect considerable savings on labor and also eliminate the possibility of cross-contamination between wells as can occur with portable pumping and sampling devices. This method is also of notable value in obtaining bacteriological samples where external sources of contamination must be avoided.

Somewhat more elaborate pumping equipment is required in small-diameter wells where the water level is below suction lift. The easiest, but not the driest, way to collect a sample is to install an airline in the well and blow the water out. However, trying to adjust airflow so that water flows smoothly over the top of the casing instead of blowing violently into the air is a difficult task. The addition of some simple, relatively inexpensive hardware to cap the well can make sampling a straightforward and easy process. Sommerfelt and Campbell (1975) have described such an installation (Figure 61)[2] and Trescott and Pinder (1970) have pumped water from as deep as 58 meters (190 feet) with this type of installation.[3] Air pressure can be supplied from a gasoline-powered air compressor, an engine air pump,

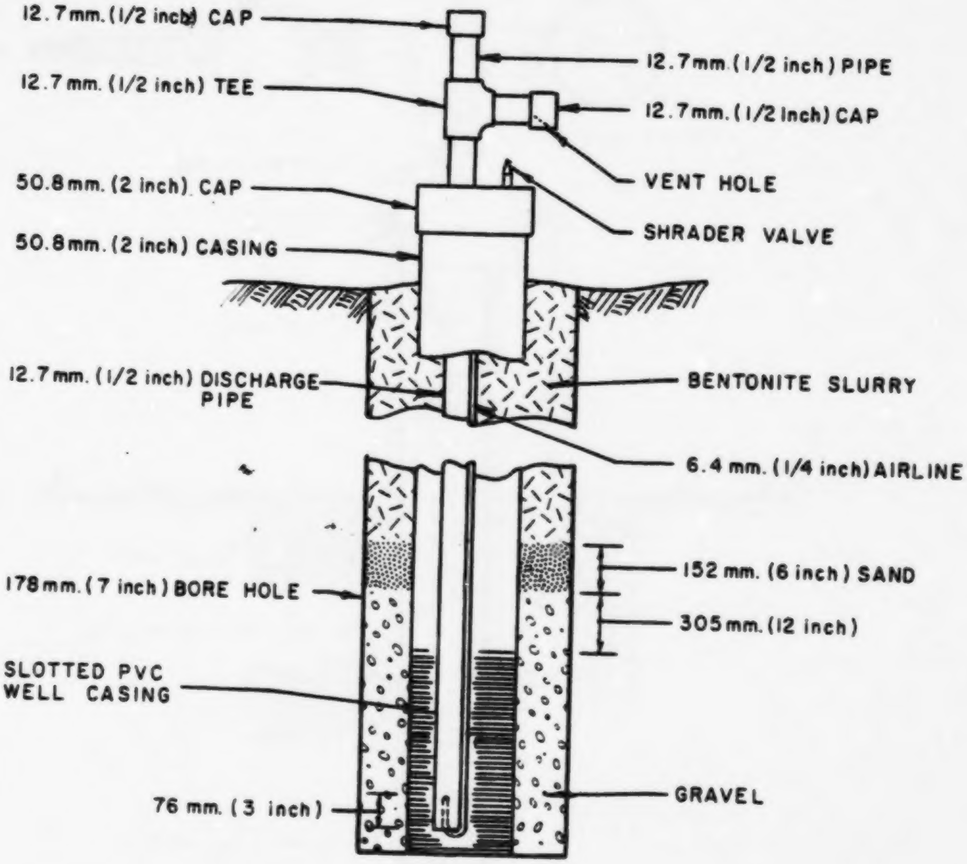

12.7mm.(1/2 inch) CAP
12.7mm.(1/2 inch) PIPE
12.7mm.(1/2 inch) TEE
12.7mm.(1/2 Inch) CAP
VENT HOLE
50.8mm.(2 inch) CAP
SHRADER VALVE
50.8mm.(2 inch) CASING
12.7mm.(1/2 inch) DISCHARGE PIPE
BENTONITE SLURRY
6.4 mm.(1/4 inch) AIRLINE
152 mm.(6 inch) SAND
178mm.(7 inch) BORE HOLE
305mm.(12 inch)
SLOTTED PVC WELL CASING
GRAVEL
76 mm.(3 inch)

Figure 60. Airlift Water Sampling Device*

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*Walker, William. Field verification of industrial hazardous material migration from land disposal sites in humid regions. U.S. Environmental Protection Agency, July, 1974.

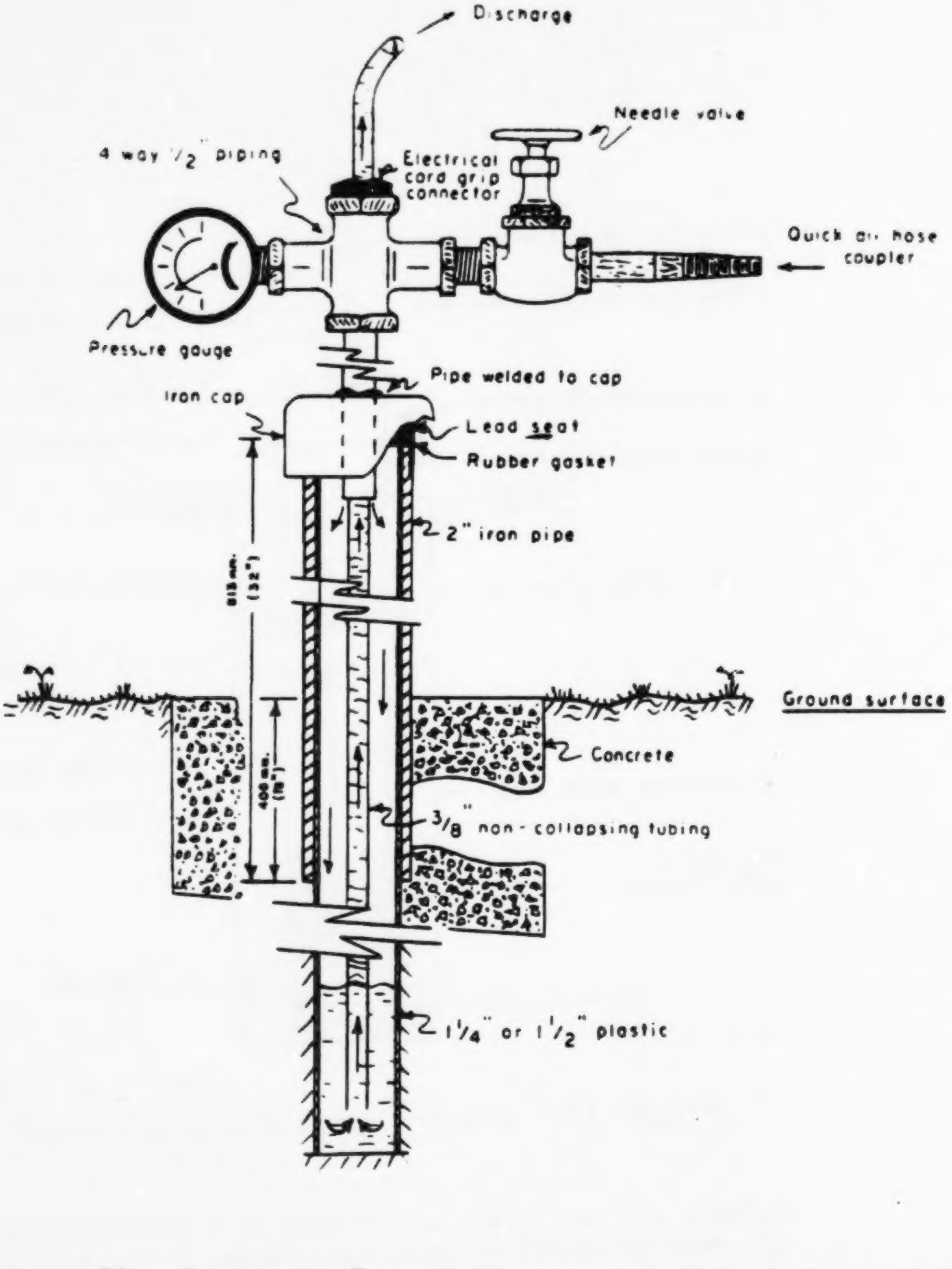
Discharge
Needle valve
4 way 1/2" piping
Electrical cord grip connector
Quick air hose coupler
Pressure gauge
Pipe welded to cap
Iron cap
Lead seat
Rubber gasket
2" iron pipe
813 mm. (32")
Ground surface
Concrete
406 mm. (16")
3/8" non-collapsing tubing
1 1/4" or 1 1/2" plastic

FIGURE 61. SCHEMATIC DIAGRAM SHOWING THE CONSTRUCTION AND MECHANISM OF AN AIRLIFT PUMP.

(After Summerfeld & Campbell, 1975)

or a compressed air cylinder. The source of air pressure selected will depend upon well site accessibility and budgetary constraints. Where a well is not accessible to vehicle transportation, a good-quality hand pump (available in stores handling racing bicycles) can be used as an air supply.

The best, but most expensive, method of collecting water samples is with a small submersible pump. McMillion and Keeley (1968) have designed a truck portable submersible pump capable of being set at depths up to 91.4 meters (300 feet) inside a casing as small as 114 mm (4½ inches) ID with a capacity of .44 to .88 l/s (7 to 14 gpm). 4 With this equipment, a pump can be set as desired in the well and operated until a representative sample can be obtained. Cross-contamination between wells from the sampling equipment will be virtually eliminated by the flushing action during pre-pumping. Provided there are no air leaks in the pumping system, samples could also be collected for dissolved gas analyses.

Samples for bacteriological examination must be collected in sterile containers. Detailed sampling procedures for bacteriological samples are given in:

- *Standard Methods*, 13th Ed., pp. 657-660;
- *Biological Analysis of Water and Wastewater*, AM 302, Millipore Corp., 1974, pp. 4-6.

Samples can be taken directly from wells with a sterile bottle in a weighted frame which can be lowered below the water surface and opened below surface. Samples can also be obtained by means of various pumping devices as previously described. However, great care must be taken to minimize sample contamination by the sampling equipment. Sample volumes of approximately 250 ml are usually satisfactory for bacteriological testing.

Sampling and preservation of samples are addressed in:

- *1973 Annual Book of ASTM Standards, Part 23, Water* and *Atmospheric Analysis*--

  pp. 72-75, "Standard Methods of Sampling Homogeneous Industrial Wastewater";

  pp. 76-91, "Standard Methods of Sampling Water".

6.2.2 Records

Adequate records should be maintained on each sample that is taken. Record information should include:

- sample description--type (ground water, surface water), volume;
- sample source--well number, location;
- sampler's identity--chain of evidence should be maintained; each time transfer of a sample occurs, a record including signatures of parties involved in transfer should be made. (This procedure can have legal significance.);
- time and date of sampling;
- significant weather conditions;
- sample laboratory number;
- pertinent well data--depth, depth to water surface, pumping schedule, and method;
- sampling method--vacuum, bailer, pressure;
- preservatives, (if any)--type and number (e.g., $NaOH$ for cyanide, H_3PO and $CuSO_4$ for phenols, etc.);
- sample containers--type, size, and number (e.g., three liter glass stoppered bottles, one gallon screw-cap bottle, etc.);
- reason for sampling--initial sampling of new landfill, annual sampling, quarterly sampling, special problem sampling in conjunction with contaminant discovered in nearby domestic well, etc.;
- appearance of sample--color, turbidity, sediment, oil on surface, etc.;
- any other information which appears to be significant--(e.g., sampled in conjunction with state, county, local regulatory authorities; samples for specific conductance value only; sampled for key indicator analysis; sampled for extended analysis; resampled following engineering corrective action, etc.);
- name and location of laboratory performing analysis;

- sample temperature upon sampling;

- thermal preservation--(e.g., transportation in ice chest);

- analytical determinations (if any) performed in the field at the time of sampling and results obtained--(e.g., pH, temperature, dissolved oxygen, and specific conductance, etc.);

- analyst's identity and affiliation.

6.2.3 Chain of Custody

Proper chain of custody procedures play a crucial role in enforcement cases. The following are some basic guidelines which have legal significance:

- As few people as possible should handle the sample.

- Stream and ground-water samples should be obtained by using standard field sampling techniques as discussed in this manual.

- The chain of custody records should be attached to the sample container at the time the sample is collected, and should contain the following information: sample number, date and time taken, source of the sample (include type of sample and name of firm), the preservative and analysis required, name of person taking sample, and the name of witness. The prefilled side of the card should be signed, timed, and dated by the person sampling. The sample container should then be sealed, containing the regulatory agency's designation, date, and sampler's signature. The seal should cover the string or wire tie of the chain of custody record, so that the record or tag cannot be removed and the container cannot be opened without breaking the seal. The tags and seals should be filled out in legible handwriting. When transferring the possession of samples, the transferee should sign and record the date and time on the chain of custody record. Custody transfers, if made to a sample custodian in the field, should be recorded for each individual sample. To prevent undue proliferation of custody records, the number of custodians in the chain of possession should be as few as possible.

If samples are delivered to the laboratory when appropriate personnel are not there to receive them, the samples should be locked in a designated area within the laboratory so that no one can tamper with them.

- Blank samples should be collected in containers, with and without preservatives, so that the laboratory analysis can be performed to show that there was no container contamination.
- A field book or log should be used to record field measurements and other pertinent information necessary to refresh the sampler's memory in the event he later becomes a witness in an enforcement proceeding. A separate set of field notebooks should be maintained for each survey and stored in a safe place where they can be protected and accounted for at all times. A standard format should be established to minimize field entries and should include the types of information listed in Section 6.2.2. The entries should then be signed by the field sampler. The responsibility for preparing and retaining field notebooks during and after the survey should be assigned to a survey coordinator or his designated representative.
- The field sampler is responsible for the care and custody of the samples collected until properly dispatched to the receiving laboratory or turned over to an assigned custodian. He must assure that each container is in his physical possession or in his view at all times or stored in a locked place where no one can tamper with it.
- Photographs can be taken to set forth exactly where the particular samples were obtained. Written documentation on the back of the photograph should include the signature of the photographer, the time, date, and site location. Photographs of this nature, which may be used as evidence, should be handled according to the established chain of custody procedures.
- Each laboratory should have a sample custodian to maintain a permanent log book in which he records for each sample the person delivering the sample, the person receiving the sample, date and time received, source of sample, sample number, how transmitted to the lab, and a number assigned to each sample by the laboratory. A standardized

> format should be established for log-book entries. The custodian should insure that heat-sensitive or light-sensitive samples or other sample materials having unusual physical characteristics or requiring special handling are properly stored and maintained. Distribution of samples to laboratory personnel who are to perform analyses should be made only by the custodian. The custodian should enter into the log the laboratory sample number, time, date, and the signature of the person to whom the samples were given. Laboratory personnel should examine the seal on the container prior to opening and should be prepared to testify that their examination of the container indicated that it had not been tampered with or opened.

6.3 SAMPLE CONTAINERS

For most samples and analytical parameters, either glass or plastic containers are satisfactory. Some exceptions are encountered such as the use of plastic for silica determinations and glass for phenols or oil and grease determination. Containers should be kept full until samples are analyzed to maintain anaerobic conditions.

As a general guide in choosing a sample container, the ideal material of construction should be non-reactive with the sample and especially the particular analytical parameter to be tested. Table 14 lists the recommended containers for various analyses.

Cleanliness of containers is of utmost importance. An effective procedure for cleaning containers is to wash with detergent, sequentially followed by: tap water rinse, nitric acid rinse, tap water rinse, hydrochloric acid rinse, tap water rinse, and finally a rinse with deionized or distilled water. In addition, the containers should be rinsed at least once with the sample at the time of sampling.

6.4 PRESERVATION OF SAMPLES AND SAMPLE VOLUME REQUIREMENTS

The following excerpt, including the tables, is a useful guide for sample preservation, sample volume requirements, and sample containers:[5]

> Complete and unequivocal preservation of samples, either domestic sewage, industrial wastes, or natural waters, is a practical impossibility. Regardless of the nature of the sample, complete stability for every constituent can never be achieved. At best, preservation techniques can only retard the chemical and biological changes that inevitably continue after the sample is removed from the parent source. The changes that take place in a sample

are either chemical or biological. In the former case, certain changes occur in the chemical structure of the constituents that are a function of physical conditions. Metal cations may precipitate as hydroxides or form complexes with other constituents; cations or anions may change valence states under certain reducing or oxidizing conditions; other constituents may dissolve or volatilize with the passage of time. Metal cations may also adsorb onto surfaces (glass, plastic, quartz, etc.) such as, iron and lead. Biological changes taking place in a sample may change the valence of an element or a radical to a different valence. soluble constituents may be converted to organically bound materials in cell structures, or cell lysis may result in release of cellular material into solution. The well-known nitrogen and phosphorus cycles are examples of biological influence on sample compositions.

Methods of preservation are relatively limited and are intended generally to (1) retard biological action, (2) retard hydrolysis of chemical compounds and complexes and (3) reduce volatility of constituents.

Preservation methods are generally limited to pH control, chemical addition, refrigeration, and freezing. Various preservatives that may be used to retard changes in samples are as follows:

Preservative	Action	Applicable to:
$HgCl_2$	Bacterial Inhibitor	Nitrogen forms, phosphorus forms
Acid (HNO_3)	Metals solvent, prevents precipitation	Metals
Acid (H_2SO_4)	Bacterial Inhibitor	Organic samples (COD, oil & grease, organic carbon)
	Salt formation with organic bases	Ammonia, amines
Alkali (NaOH)	Salt formation with volatile compounds	Cyanides, organic acids
Refrigeration	Bacterial Inhibitor	Acidity-alkalinity, organic materials, BOD, color, odor, organic P, organic N, carbon, etc., biological organism (coliform, etc.)

In summary, refrigeration at temperatures near freezing or below is the best preservation technique available, but it is not applicable to all types of samples.

The recommended choice of preservatives for various constituents is given in Table 14 (sic). These choices are based upon the accompanying references and on information supplied by various Regional Analytical Quality Control Coordinators.

TABLE 14

RECOMMENDATION FOR SAMPLING AND PRESERVATION OF SAMPLES ACCORDING TO MEASUREMENT[1]

Measurement	Vol. Req. (ml)	Container	Preservative	Holding Time (6)
Acidity	100	P,G(2)	Cool, 4°C	24 Hrs.
Alkalinity	100	P,G	Cool, 4°C	24 Hrs.
Arsenic	100	P,G	HNO_3 to pH<2	6 Mos.
BOD	1,000	P,G	Cool, 4°C	6 Hrs.(3)
Bromide	100	P,G	Cool, 4°C	24 Hrs.
COD	50	P,G	H_2SO_4 to pH<2	7 Days
Chloride	50	P,G	None Req.	7 Days
Chlorine Req.	50	P,G	Cool, 4°C	24 Hrs.
Color	50	P,G	Cool, 4°C	24 Hrs.
Cyanides	500	P,G	Cool, 4°C NaOH to pH 12	24 Hrs.
Dissolved Oxygen				
Probe	300	G, only	Det. on site	None
Winkler	300	G, only	Fix. on site	None
Fluoride	300	P,G	Cool, 4°C	7 Days
Hardness	100	P,G	Cool, 4°C	7 Days

TABLE 14 (continued)

Measurement	Vol. Req. (ml)	Container	Preservative	Holding Time (6)
Iodide	100	P,G	Cool, 4°C	24 Hrs.
MBAS	250	P,G	Cool, 4°C	24 Hrs.
Metals				
Dissolved	200	P,G	Filter on site HNO_3 to $pH<2$	6 Mos.
Suspended			Filter on site	6 Mos.
Total	100		HNO_3 to $pH<2$	6 Mos.
Mercury				
Dissolved	100	P,G	Filter HNO_3 to $pH<2$	38 Days (Glass) 13 Days (Hard Plastic)
Total	100	P,G	HNO_3 to $pH<2$	38 Days (Glass) 13 Days (Hard Plastic)
Nitrogen				
Ammonia	400	P,G	Cool, 4°C H_2SO_4 to $pH<2$	24 Hrs.(4)
Kjeldahl	500	P,G	Cool, 4°C H_2SO_4 to $pH<2$	24 Hrs.(4)
Nitrate	100	P,G	Cool, 4°C H_2SO_4 to $pH<2$	24 Hrs.(4)
Nitrite	50	P,G	Cool, 4°C	24 Hrs.(4)

TABLE 14 (continued)

Measurement	Vol Req. (ml)	Container	Preservative	Holding Time (6)
NTA	50	P,G	Cool, 4°C	24 Hrs.
Oil & Grease	1,000	G only	Cool, 4°C H_2SO_4 to pH<2	24 Hrs.
Organic Carbon	25	P,G	Cool, 4°C H_2SO_4 to pH<2	24 Hrs.
pH	25	P,G	Cool, 4°C Det. on site	6 Hrs.(3)
Phenolics	500	G only	Cool, 4°C H_3PO_4 to pH<4 1.0 g $CuSO_4$/1-	24 Hrs.
Phosphorus				
Ortho-phosphate, Dissolved	50	P,G	Filter on site Cool, 4°C	24 Hrs. (4)
Hydrolyzable	50	P,G	Cool, 4°C H_2SO_4 to pH<2	24 Hrs.(4)
Total	50	P,G	Cool, 4°C	24 Hrs.(4)
Total, Dissolved	50	P,G	Filter on site Cool, 4°C	24 Hrs.(4)
Residue				
Filterable	100	P,G	Cool, 4°C	7 Days
Non-filterable	100	P,G	Cool, 4°C	7 Days
Total	100	P,G	Cool, 4°C	7 Days
Volatile	100	P,G	Cool, 4°C	7 Days
Setteable Matter	1,000	P,G,	None Req.	24 Hrs.
Selenium	50	P,G	HNO_3 to pH<2	6 Mos.

TABLE 14 (continued)

Measurement	Vol Req. (ml)	Container	Preservative	Holding Time (6)
Silica	50	P only	Cool, 4°C	7 Days
Specific Conductance	100	P,G	Cool, 4°C	24 Hrs.(5)
Sulfate	50	P,G	Cool, 4°C	7 Days
Sulfide	50	P,G	2 ml zinc acetate	24 Hrs.
Sulfite	50	P,G	Cool, 4°C	24 Hrs.
Temperature	1,000	P,G	Det. on site	None
Threshold odor	200	G only	Cool, 4°C	24 Hrs.
Turbidity	100	P,G	Cool, 4°C	7 Days

1. More specific instructions for preservation and sampling are found with each procedure as detailed in this manual. A general discussion on sampling water and industrial wastewater may be found in ASTM, Part 23, p. 72-91 (1973).

2. Plastic or Glass

3. If samples cannot be returned to the laboratory in less than 6 hours and holding time exceeds this limit, the final reported data should indicate the actual holding time.

4. Mercuric chloride may be used as an alternate preservative at a concentration of 40 mg/l, especially if a longer holding time is required. However, the use of mercuric chloride is discouraged whenever possible.

5. If the sample is stabilized by cooling, it should be warmed to 25°C for reading or temperature correction made and results reported at 25°C.

6. It has been shown that samples properly preserved may be held for extended periods beyond the recommended holding time.

Additional useful information relative to preservation of polluted waters, wastewaters. etc., is available in Standard Methods, 13th Ed. 1971, pp. 368-369.[5]

Furthermore, Standard Methods provides a very useful "Sampling and Storage" section for many of the analytical methods, offered.

6.5 PRESERVATION OF SAMPLES IN THE FIELD

Samples should be preserved at low temperatures during transport to the laboratory for analysis. A convenient method is to use an insulated cooler containing ice so that a temperature of 0° to 10°C is maintained.

If possible, appropriate chemical preservation should be performed in the field for various analytical parameters at the time of sampling. In this case, separate bottles and chemical preservatives are required for particular parameters. As an example, for the extended analyses group in Chapter 4, proper preservative techniques would require splitting the sample into approximately 10 bottles. For this reason, sampling a large number of wells for several analyses can become a cumbersome exercise. Regardless of the method of preservation, analyses should be performed as soon after sampling as is practicably possible.

REFERENCES

1. Effective use of high water table areas for sanitary landfill. Final report (SW-57d.1). U.S. Environmental Protection Agency, 1973.

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3. Trescott, P.C., and G.F. Pinder. Air pump for small-diameter piezometers. Ground Water, 8(3):10-15, 1970.

4. McMillon, L.G. and J.W. Keeley. Sampling equipment for ground-water investigations. Ground Water, 6(2):9-11, 1968.

5. Methods for chemical analysis of water and waste. Report No. 625/6-74-003. Environmental Protection Agency, pp. vi-xii.

6. Standard methods for the examination of water and wastewater, 13th Edition. American Public Health Association, 1970.

7.0 ANALYTICAL METHODS

7.1 INTRODUCTION

Reliable, cost-effective analytical methods must be selected and applied in order to successfully implement Basic Indicator, and Extended Analysis programs.

The parameters of interest in the analytical characterizations of leachate are usually physical, chemical, and biological. Normally, the desired information is quantitative rather than qualitative, although qualitative data may be required at times for special problems. For purposes of this manual, consideration will be given only to the quantitative aspects of the analytical data.

As stated previously in Chapter 6, "Leachate represents an extremely complex system containing soluble, insoluble, organic, inorganic, ionic, nonionic, and bacteriological constituents in an aqueous medium. Actual types, numbers, and levels of constituents are widely variable..."

When dealing with a complex material of variable composition, such as leachate, it is recognized that there is a serious potential for numerous interferences in the determination of a given parameter.

The physical measurements, such as specific conductance and pH, are not normally subject to appreciable interference; but many of the chemical and biological determinations are readily affected by matrix interferences.

When an analyst wishes to perform a quantitative determination on a particular parameter, he must decide which analytical method will be used. There is usually a choice among several standard methods which can be applied to a given determination. Among the many factors which must be considered in the choice of an analytical method are the following:

- sensitivity, precision, and accuracy required;
- nature of the matrix and its effect upon the determination (interferences);
- available equipment, manpower and instrumentation;

- quantity of sample available;
- level of expertise of the analyst;
- number of samples to be analyzed;
- turn-around time;
- history and available information regarding the sample;
- reason for performing the analysis;
- how the analytical data will be applied;
- other parameters, if any, to be determined on the sample;
- cost factors.

When all pertinent considerations of this nature have been carefully weighed, the decision is then made to apply a particular standard method to the problem. There are several literature sources of standard analytical methods which can be applied, either directly or with modification, to the analysis of leachate samples. Because of their importance in analytical procedures, the following three publications are herein referenced as the "rule books" for the analysis of leachate samples:

- *Standard Methods for the Examination of Water and Wastewater*, 13th Ed., APHA, 1971.
- *Manual of Methods for Chemical Analysis of Water and Wastes*, U. S. Environmental Protection Agency (EPA), 1974.
- 1973 *Annual Book of ASTM Standards*, Part 23, Water; Atmospheric Analysis.

The following quotation is relative to the analysis of polluted waters and other similar samples in *Standard Methods for the Examination of Water and Wastewater*, P. 367:*

> These procedures described in Part 200 of this manual are intended for the physical and chemical examination of wastewaters of both domestic and industrial origin, treatment plant effluents, polluted waters, sludges and bottom sediments. An effort has been made to present

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*Due to the magnitude of importance of quoted material within this chapter, references are cited with the text rather than at the end of the chapter.

> methods which apply as generally as possible and to indicate modifications which are required for samples of unusual composition, such as certain industrial wastes. However, because of the wide variety of industrial wastes, the procedures given here cannot cover all possibilities and may not be suitable for all wastes and combinations of wastes. Hence, some modification of a procedure may be necessary in specific instances. Whenever a procedure is modified, the nature of the modifications must be plainly stated in the report of results. The procedures which are indicated as being intended for the examination of sludges and bottom sediments may not apply without modification to chemical sludges or slurries.

In this same vein, the following comments are made in Handbook for Analytical Quality Control in Water and Wastewater Laboratories, U. S. EPA, 1972, p. 1-3:

> Regardless of the analytical method used in the laboratory, the specific methodology should be carefully documented. In some water pollution reports it is customary to state that Standard Methods have been used throughout. Close examination indicates, however, that this is not strictly true. In many laboratories, the standard method has been modified because of recent research or personal preferences of the laboratory staff. In other cases, the standard method has been replaced with a better one. Statements concerning the methods used in arriving at laboratory data should be clearly and honestly stated. The methods used should be adequately referenced and the procedures applied exactly as directed.
>
> Knowing the specific method which has been used, the reviewer can apply the associated precision and accuracy of the method when interpreting the laboratory results. If the analytical methodology is in doubt, the data user may honestly inquire as to the reliability of the result he is to interpret.
>
> The advantages of strict adherence to accepted methods should not stifle investigations leading to improvements in analytical procedures. In spite of the value of accepted and documented methods, occasions do arise when a procedure must be modified to eliminate unusual interference, or to yield increased sensitivity. When modification is necessary, the revision should be carefully worked out to accomplish the desired result. It is advisable to assemble data using both the regular and the modified procedure to show the superiority of the latter. This useful information can be brought to the

attention of the individuals and groups responsible for methods standardization. For maximum benefit, the modified procedure should be rewritten in the standard format so that the substituted procedure may be used throughout the laboratory for routine examination of samples. Responsibility for the use of a non-standard procedure rests with the analyst and his supervisor, since such use represents a departure from accepted practice.

7.2 ALTERNATE ANALYTICAL METHODS

7.2.1 Method Comparability

Relative to the use of alternate analytical methods for the National Pollution Discharge Elimination System, the EPA has published guidelines in the Federal Register, October 16, 1973, as follows:

> Typical Comparability Testing Procedure. (sic)
>
> This procedure is designed to provide data on the comparability (equivalency) of two dissimilar analytical methods for measurement of the same property or constituent.
>
> In making the comparison, one method is assumed to be satisfactory (standard) and the second or alternate method is compared for equivalency. To provide sufficient data to apply statistical measurements of significance, the following determinations are required:
>
> 1. Using an effluent sample representative of normal operating processes, well-mixed between aliquot withdrawal, run seven replicate determinations by each method.
>
> Report values in the following manner:
>
> TABLE 1
>
> Effluent Sample Representative of Normal Operating Conditions
>
> | Aliquot | Standard Method* | Alternate Method* |
> |---|---|---|
> | List 1 through 7 | | |
>
> *Cite method reference
>
> 2. If variations occur in the concentration of the measured constituent in the plant effluent, repeat the above testing on two more samples, one collected

at the highest level of constituent normally encountered in the waste samples examined by the laboratory and one having a concentration at or near the lowest level usually examined. Report values in the following manner:

TABLE 2

Effluent Samples of Varying Composition

Aliquot	Low Level	High Level
List 1 through 7		

3. Using the sample from 1, add a small volume of standard solution sufficient to double the concentration. Run 7 replicate determinations by each method. Report values as Table 3:

Effluent Samples Plus Standard Solution, in the same way as Table 1. Cite source and amount of standard solution; it should be proportioned to the original concentration. The above procedure must be followed on each outfall for which a permit is issued, unless it can be shown that the outfalls in question are comparable.

A comparability test procedure for analytical methods used on landfill leachate samples can be modelled after the above-cited EPA procedure. Samples, instead of representing plant effluents, will represent potentially leachate-enriched ground and/or surface waters. The results of the standard and alternate methods should be compared for statistically significant differences. If the alternate method proves to be equal to or better than the standard method, it should be considered an acceptable analytical method for the determination of the particular parameter in the leachate sample.

7.2.2 Additional Analytical Methods

A considerable amount of valuable pertinent information on analytical methodology and data is available in Standard Methods for the Examination of Water and Wastewater, 13th Edition, 1971. The particular subjects of interest in the analysis of leachate samples are:

- Other (instrumental) methods of analysis, including Atomic Absorption Spectroscopy, Flame Photometry, Emission Spectroscopy, Polarography, Potentiometric Titration, Specific Ion Electrodes and Probes, Gas Chromatography, and Automated Analytical Instrumentation. (pp. 12-15)

- Interferences and methods used for their elimination. (pp. 15-18)
- Expression of Results. (pp. 18-20)
- Significant Figures. (pp. 20-21)
- Precision and Accuracy, Statistical Approach, Standard Deviation, Range, Rejection of Experimental Data, Presentation of Precision and Accuracy Data, Quality Control. (pp. 22-25)
- Graphical Representation of Data, Method of Least Squares. (pp. 25-27)
- Self-evaluation (desirable philosophy for the analyst) (p. 27)
- Methods Evaluation by the Committee on Standard Methods of the Water Pollution Control Federation. (pp. 369-370)

7.3 SPECIFIC ANALYTICAL METHODS FOR THE ANALYSIS OF RELATIVELY CONCENTRATED LEACHATE SAMPLES

7.3.1 Introduction

Specific analytical methods for the analysis of relatively concentrated leachate samples were investigated in the report "Compilation of Methodology for Measuring Pollution Parameters of Landfill Leachate" by E.S.K. Chian and F.B. DeWalle, University of Illinois, EPA Program Element No. 1DB064. It is stated in the abstract, P. IV of the subject report:

> Since different analytical methods can be used to determine a specific parameter, a preliminary laboratory evaluation was made of those methods least subject to interferences. All analyses were conducted with a relatively concentrated leachate sample obtained from a lysimeter filled with milled solid waste. The results indicate that strong interferences are sometimes encountered when using colorimetric tests due principally to the color and suspended solids present in leachate. In such instances, alternative methods were evaluated or recommendations were made to reduce the interfering effects. Automated chemical analysis using colorimetric methods can sometimes experience significant interferences.
>
> Further research is necessary to evaluate additional methods using leachate samples of different strengths and collected from landfills of different ages. The precision and sensitivity of each method will also

> have to be determined. The interfering parameter should be quantified to allow predictions of its magnitude with leachate samples of different strengths.

Also in the above-cited report, Introduction, p. 3, it is stated:

> It is the purpose of the present study to review the analytical methods to determine contaminants as reported in the literature. The methods compiled and evaluated in this study were generally reported in the literature; additional information was obtained by contacting the principal investigators. Interferences in the chemical analysis due to the complex nature of the leachate as enumerated in the reported studies are listed in this report.
>
> The compilation showed that different methods subject to different interferences are used to determine a certain parameter. For each parameter, only that method was evaluated in this laboratory which was found to have the smallest interference. The laboratory evaluation tested the method for its susceptibility to certain interferences commonly found in leachate. In addition, the accuracy of the method was tested. All laboratory analyses were performed using a high strength leachate sample obtained from a recently installed lysimeter filled with milled refuse. Recommendations made in this report, therefore, only apply to leachate of similar strength. No evaluation was made of precision and sensitivity of each method since this was beyond the scope of the work. Realizing the above restrictions, recommendations were made in the present study for the selection of those methods least subject to interference. Further recommendations were made concerning modifications of the selected methods.

7.3.2 Measurement of Interference Effects

Two general procedures were used by Chian and DeWalle to deal with interference effects in the evaluation of specific analytical methods. These procedures were the Standard Addition Method and Dilution Method. These methods are discussed in this report in pp. 12-15.

In general, it would be expected that interferences encountered in concentrated leachates would be relatively severe and constitute "worst case" effects when compared with more dilute leachates. Leachates obtained in the field (landfills) for analysis may vary greatly in total concentration; i.e., from total concentration of minimum detectability to a highly concentrated product. The ratios of the individual contaminants present in the leachate are also variable and must be considered when evaluating interference effects in a given analytical method.

The analyst, therefore, must always evaluate a specific analytical method relative to a specific leachate sample. General guidelines for handling interferences are of great value. The judgment of the analyst is of prime importance in applying the guidelines to the specific problems at hand. Experience with a given leachate is obviously of practical value. The degree of accuracy, sensitivity, and precision required in a specific analytical problem will constitute foremost considerations in the final selection of the method and possible modifications.

7.4 ANALYTICAL METHODS

In discussing individual analytical methods in their aforementioned report (pp. 16-121) Chian and DeWalle address the following aspects in each case: Principle, Interferences, Previous Studies, Evaluation of the Method, Recommendations, and Procedures.

The methods discussed in the report are as follows:

- Physical Parameters: pH, Oxidation Reduction Potential (ORP), Specific Conductance, and Residue.
- Organic Chemical Parameters: C.O.D., T.O.C., Volatile Acids, Tannin and Lignin, Organic Nitrogen.
- Inorganic Chemical Parameters: Chloride, Sulfate, Phosphate, Alkalinity and Acidity, Nitrate, Nitrite, Ammonia, Sodium and Potassium, Calcium and Magnesium, Hardness, Heavy Metals.
- Biological Parameters: B.O.D., Coliform Bacteria (Total and Fecal).
- Miscellaneous Determinations.

The report also contains a useful appendix of parameters and methods used by various investigators. (Appendix A, p. 125 - Survey of physical, chemical and biological methods used by various investigators).

7.5 BRIEF DESCRIPTION OF SPECIFIC ANALYTICAL METHODS FOR LEACHATE ANALYSIS

Following is a brief description of the analytical methods as recommended by Chian and DeWalle for the analysis of concentrated leachate. Information on interferences and methods of minimizing them, is included.

1. Physical Parameters:

A. pH Determination:

Method: Electrometric determination using a glass indicating electrode and calomel reference electrode or a combination electrode. The procedure is according to Standard Methods, p. 279.

Interference: The glass electrode is relatively immune to interference from color, turbidity, colloidal matter, free chlorine, oxidants, or reductants as well as from high saline content, except for a sodium error at high pH. The error caused by high sodium ion concentrations at a pH above 10 may be reduced by using special "low-sodium error" electrodes.

B. Oxidation Reduction Potential (ORP) Determination:

Method: The measurement is made with a pH meter, using a platinum indicating electrode and a calomel reference electrode. The pH determination is made concurrently.

Interference: Oxidation of the sample and presence of foreign matter on electrodes can cause erroneous results.

C. Specific Conductance Determination:

Method: The determination is performed with a commercially available meter and an electrode with a cell constant of 1.0. Both temperature and pH are determined concurrently as they affect the results. Reference is made to Standard Methods, pp. 326-327.

Interference: Fouling of the electrode surfaces may occur and checks of the indicated conductance using another cell or standard solution may be necessary. A chromium-sulfuric acid mixture is effective in cleaning the electrodes.

D. Residue Determination:

Total solids is determined after drying to constant weight at 103-105°C and the volatile solids is determined from the weight loss at 550°C for one hour. The suspended solids (filterable residue) is determined using a glass fiber filter and drying to constant weight at 103-105°. The following reference is given: Standard Methods, pp. 289, 292, 293.

Interference: The determination of the total solids can vary due to volatilization of part of the organic matter, loss of occluded water and gases from heat-induced chemical decomposition. The suspended solids determination is affected by the physical nature of the material in suspension, the pore size of the filter, and the area and thickness of the mat.

2. Organic Chemical Parameters:

A. Chemical Oxygen Demand (C.O.D.)

Method: The C.O.D. determination is performed by the dichromate reflux method, according to Standard Methods, pp. 496-499. If the C.O.D. is less than 100 mg/liter, more accurate results may be obtained by using the low-level C.O.D. procedure cited in the same reference.

Interference: Straight-chain aliphatic compounds, aromatic hydrocarbons, and pyridine are not oxidized to any appreciable extent, although this method gives more nearly complete oxidation than the permanganate method. The straight-chain compounds are more effectively oxidized when silver sulfate is added as a catalyst; however, silver sulfate reacts with chlorides, bromides, or iodides to produce precipitates which are only partially oxidized by the procedure. There is no advantage in using the catalyst in the oxidation of aromatic hydrocarbons, but it is essential to the oxidation of straight-chain alcohols and acids.

The oxidation and other difficulties caused by the presence of chlorides in the sample may be overcome by employing the following method which is a complexing technique for the elimination of chlorides from the reaction. This is accomplished by adding mercuric sulfate to the samples before refluxing. This ties up the chloride ion as a soluble mercuric chloride complex, which greatly reduces its ability to react further.

B. Total Organic Carbon (T.O.C.)

Method: The T.O.C. analysis is performed by the combustion-infrared method according to Standard Methods, pp. 257-259.

Interference: The removal of carbonate and bicarbonate by means of acidification and purging with nitrogen gas can result in the loss of very volatile organic substances. Another important loss can occur from the failure of large carbon-containing particles in the sample to enter the hypodermic needle used for injection.

C. Volatile Acids (Total Organic Acids)

Method: Volatile acids are determined by the column-partition chromatographic method involving extraction from the column with chloroform-butanol solvent and titration with 0.02N standard sodium hydroxide titrant, as given in Standard Methods, pp. 577-580. Standard amounts of acetic acid are added to determine the recovery of the method.

Interference: The chloroform-butanol solvent system employed in the method is capable of eluting organic acids other than the volatile acids and some synthetic detergents as well. Crotonic, adipic, pyruvic, phthalic, fumaric, lactic, succinic, malonic, gallic, aconitic, and oxalic acids, alkyl sulfates and alkyl-aryl sulfonates are all adsorbed by silicic acid and eluted when present.

D. Tannin and Lignin

Method: Aromatic hydroxyl groups reduce tungstophosphoric and molybdophosphoric acids to form a blue color which is measured spectrophotometrically at a wavelength of 700nm.

Interference: Other easily oxidizable substances such as reduced metal ions, sulfides, and nitrite may give a similar reaction in this test and caution should therefore be exercised in evaluating the results. The largest interference is caused by ferrous iron and 2 mg/liter will give a color equivalent to 1 mg/liter tannic acid.

E. Organic Nitrogen

Method: In the presence of sulfuric acid, potassium sulfate and mercuric sulfate catalyst, the amino nitrogen of many organic materials is converted to ammonium bisulfate. After the mercury ammonium complex in the digestate has been decomposed by sodium thiosulfate, the ammonia is distilled from an alkaline medium and absorbed in boric acid. The ammonia is determined colorimetrically or, if preferred, by titration with a standard mineral acid. The sensitivity of the colorimetric methods makes them useful for the determination of organic nitrogen levels below 1 mg/liter. The titrimetric method of measuring the ammonia in the distillate is suitable for the determination of a wide range of organic nitrogen concentrations depending upon the volume of boric acid absorbent used and the concentration of the standard acid titrant. Should the total kjeldahl nitrogen and ammonia nitrogen be determined individually, the "organic nitrogen" can then be obtained by difference.

Interference: The only interferences which are encountered may be due to the incomplete oxidation of the organic nitrogen to ammonia. The organic nitrogen determination is according to Standard Methods, pp. 244-248.

3. Inorganic Chemical Parameters:

A. Chloride:

Method: In biologically stabilized leachate samples in which color does not cause any interference, the chloride determination is conducted with the mercuric nitrate titration method given in Standard Methods, pp. 97-99.

Interference: Bromide and iodide are titrated with mercuric nitrate in the same manner as chloride. Chromate, ferric, and sulfite ions interfere when present in excess of 10 mg/liter.

Alternate Method: In strongly polluted leachate, chloride ion is determined by the potentiometric titration method with silver nitrate solution using a glass and silver-silver chloride electrode system according to Standard Methods, pp. 377-380.

Interference: Iodide and bromide also are titrated as chloride. Ferricyanide causes high results and must be removed. Chromate and dichromate interfere and should be reduced to the chromic state or removed. Ferric iron interferes if present in an amount that is substantially higher than the amount of chloride. Chromic ion, ferrous iron, and phosphate do not interfere.

Grossly contaminated samples usually require pre-treatment. Where contamination is minor, some contaminants can be destroyed simply by the addition of nitric acid.

B. Sulfate:

Method: Sulfate is precipitated in a hydrochloric acid medium as barium sulfate by the addition of barium chloride. The precipitation is carried out near the boiling temperature; and after a period of digestion, the precipitate is filtered, washed with water until free of chlorides, dried at 105°C, and weighed as $BaSO_4$ according to Standard Methods, pp. 331-333.

Interferences leading to high results: Suspended matter, silica, barium chloride precipitant, nitrate, sulfite, and water are the principal factors in positive errors. Suspended matter may be present in both the sample and the precipitating solution; soluble silicate may be rendered insoluble and sulfite may be oxidized to sulfate during processing of the sample. Barium nitrate, barium chloride and water are occluded to some extent with the barium sulfate.

Interferences leading to low results: Alkali and metal sulfates frequently yield low results. This is especially true of alkali hydrogen sulfates. Occlusion of alkali sulfate with barium sulfate causes the substitution of an element of lower atomic weight than barium in the precipitate. Hydrogen sulfates of alkali metals act similarly and, in addition, decompose on being heated. Heavy metals, such as chromium and iron, cause low results by interfering with the complete precipitation of sulfate and by formation of heavy metal sulfates. Barium sulfate has small but significant solubility, which is increased in the presence of acid. Although an acid medium is necessary to prevent precipitation of barium carbonate and phosphate, it is important to limit its concentration to minimize the solution effect.

C. Phosphate:

Method: The total phosphorus content of the sample includes all of the orthophosphates and condensed phosphates, both soluble and insoluble, and organic and inorganic species. To release phosphate from combination with organic matter, a digestion or oxidation technic is called for. The rigor of the digestion required depends upon the type of sample. Three digestion technics, in order of decreasing rigor, are perchloric acid digestion, sulfuric acid digestion, and persulfate digestion. Following digestion, the liberated orthophosphate is determined colorimetrically. The colorimetric method used, rather than the digestion procedure, governs in matters of interference and minimum detectable concentration.

The ascorbic acid colorimetric method is used to measure total phosphorus concentration in leachate, using the persulfate digestion. The amount of recommended persulfate digestion reagent is 400 mg/100 ml sample, while the digestion time recommended by Standard Methods is sufficient to hydrolyze the phosphorus. The ortho-phosphate test as determined by the ascorbic acid method does not experience significant interference and should be run on the anaerobically stored leachate after as little dilution as possible. In order to obtain reliable results, a standard addition or progressive dilution curve should be established for the total phosphorus determination. Such steps are not necessary for the orthophosphate determination. References for the total phosphate and ascorbic acid methods are in Standard Methods, pp. 524-526 and 532-534.

Interference: Arsenates react with the molybdate reagent to produce a blue color similar to that formed with phosphate. Concentrations as low as 0.10 mg/liter arsenic interfere with the phosphate determination. Hexavalent chromium and nitrite interfere to give results about 3% low at concentrations of 1.0 mg/liter and 10-15% low at concentrations of 10 mg/liter chromium and nitrite. Sulfide (Na_2S) and silicate do not interfere in concentrations of 1.0 and 10.0 mg/liter.

D. Alkalinity and Acidity:

Method: The alkalinity of a water is the capacity to accept protons and is determined by titrating the sample with strong mineral acid to a specified endpoint. The acidity of a water is the capacity to donate protons and is determined by titrating the sample using strong mineral base to a specified endpoint. In unpolluted waters, Standard Methods recommends titration endpoints at pH 8.3 and between 5.1 and 4.5, depending upon the magnitude of the alkalinity. In polluted waters, it recommends an arbitrary endpoint of 8.3 (phenolphthalein endpoint) and 3.7 (methyl orange endpoint) unless a potentiometric titration curve indicates a distinct inflection point which can be employed, and the endpoint should then be specified with the analysis. From the titration curve, the inflection point can be determined, whereafter the amount of acid or base, expressed as $CaCO_3$ equivalent, can be calculated. The potentiometric titration is also to be used for colored or turbid waters where difficulties arise in determining a color indicator endpoint.

It is recommended that alkalinity and acidity determinations be made potentiometrically on the undiluted samples and that the endpoints be used as determined from the titration curve. These endpoints should be specified in reporting the results of the analysis. Standard Methods references: pp. 54-56 and 370-375.

Interference: Standard Methods indicates that the acidity and alkalinity determination is not restricted only to bicarbonate species but is also contributed by other weak acids or bases and hydroxides. The test, therefore, only indicates general properties of the leachate and cannot be used for the quantitative determination of specific species.

E. Nitrate:

Method: Nitrate is determined with the specific ion electrode or by the brucine-sulfanilic acid colorimetric method. Standard Methods reference for the colorimetric method is pp. 461-464.

Interference: It is preferable to use an undiluted sample when measuring nitrates with the specific ion electrode. Standard amounts of nitrate should be added to the sample to determine the recovery of the method.

In the brucine colorimetric method, all strong oxidizing or reducing agents interfere. The presence of oxidizing agents may be determined by the addition of orthotolidine reagent, as in the measurement of residual chlorine. Ferrous and ferric iron and quadrivalent manganese give slight positive interferences; but in concentrations less than 1 mg/liter, these are negligible. The interference due to nitrite up to 0.5 mg NO_2-N/1 is eliminated by the use of sulfanilic acid. Chloride interference is masked by the addition of excess NaCl. High concentrations of organic matter, such as in undiluted raw wastewater will usually interfere. Suspended solids and color may be removed with a massive lime dosage of 5,000 to 10,000 mg/liter $Ca(OH)_2$. Aluminum hydroxide is not as effective as a coagulant.

F. Nitrite:

Method: Nitrite is determined colorimetrically at 520 nm through the formation of a reddish-purple azo dye produced at pH 2.0 to 2.5 by the coupling of diazotized sulfanilic acid with naphthylamine hydrochloride. The procedure is given in Standard Methods, pp. 240-243. The naphthylamine is replaced by n-(naphthyl) ethylenediamine dihydrochloride. Standard amounts of nitrite nitrogen are added to the filtered sample.

Interference: The following ions interfere due to precipitation under the conditions of the test and, therefore, should be absent: antimonous, auric, bismuth, ferric, lead, mercurous, silver, chloroplatinate, and metavanadate. Cupric ion may cause low results by catalyzing the decompositon of the diazonium salt. Colored ions which alter the color system should likewise be absent. When small amounts of suspended solids seriously impair nitrite recovery, a sample may be passed through a membrane filter (0.45μ pore size) to achieve the necessary clarification before color development is undertaken. The determination should be made promptly on fresh samples to prevent bacterial conversion of the nitrite to nitrate or ammonia.

G. Ammonia:

Method: Two methods are recommended for determination of ammonia. One uses the selective ion ammonia electrode with sufficient sample dilution to reduce matrix interference of the leachate (Ref.: U. S. EPA Methods for Chemical Analysis of Water and Wastes, 1974, pp. 165-167). The other method uses distillation followed by titration of the ammonia in the distillate with standard 0.02N H_2SO_4 with mixed methyl red-methylene blue as the indicator. For this method, a maximum concentration of 75 mg/liter ammonia in the diluted sample is recommended, unless additional buffer is used. A pH of 7.4 is sufficiently high to distill off the ammonia. A pH of 9.4 is too high, causing partial destruction of the organic nitrogen. (Ref.: Standard Methods, pp. 224-226 and 246-247)

Interference:

Selective ion electrode method: Color and turbidity have no effect on the measurements and distillation is not necessary. Volatile amines act as a positive interference. Mercury interferes by forming a strong complex with ammonia. Thus, the samples cannot be preserved with mercuric chloride.

Distillation-titration method: The largest difficulty in the distillation step is maintaining the pH of 7.4. When the pH of the distillation mixture is too high, certain organic nitrogen compounds are converted to ammonia, thus increasing the apparent concentration, and when the pH is too low, the recovery of ammonia is too low. Ammonia recovery from preliminary distillation will be low on water samples containing more than 250 mg/liter calcium unless the pH is properly adjusted before distillation is undertaken. The calcium and the phosphate buffer react to precipitate calcium phosphate, releasing hydrogen ions and lowering the pH. The titrimetric procedure is also subject to amine interference because the standard acid can react with such alkaline bodies. However, the titration procedure is free of interference from neutral organic compounds.

H. Sodium and Potassium:

Method: The most rapid and sensitive method uses flame emission photometry to determine sodium and potassium concentrations at wavelengths of 589.0 nm and 766.5 nm, respectively (Ref.: Standard Methods, Sodium: pp. 316-320 and Potassium: pp. 283-284).

Sodium and potassium may also be determined by atomic absorption spectroscopy (Ref.: U. S. EPA Methods for Chemical Analysis, etc., Sodium: pp. 147-148 and Potassium pp. 143-144).

Interferences:

Flame photometry: Particulate matter can cause burner clogging and should be removed by filtration through a medium pore-sized filter paper.

Potassium and calcium have been reported to interfere with the sodium determination by the internal-standard method if the potassium-to-sodium ratio is 5:1 or greater and the calcium-to-sodium ratio is 10:1 or higher. When these ratios are exceeded, the calcium and potassium should be determined first so that the approximate concentration of interfering ions may be added, if necessary, to the sodium calibration standards. Magnesium interference does not appear until the magnesium-to-sodium ratio exceeds 100, a rare occurrence. Among the common anions capable of causing radiation interference are chloride, sulfate and bicarbonate in relatively large amounts.

Atomic absorption spectroscopy: An extensive evaluation of the method showed the necessity to add the easily ionizable cesium ion to the sample at a concentration of 1,000 mg/liter in the final dilution, to suppress ionization of the analyte ion in the determination of sodium and potassium. Sodium may interfere in the potassium determination if present at much higher levels than the potassium. This effect can be compensated by approximately matching the sodium content of the potassium standards with that of the sample.

I. Calcium and Magnesium:

Method: Calcium and magnesium are determined by atomic absorption spectroscopy, using 10,000 mg/liter lanthanum to reduce interference. (Ref.: Standard Methods, pp. 212-213 and U. S. EPA Methods for Chemical Analysis, Calcium pp. 103-104 and Magnesium pp. 114-115).

Interference: Phosphate and sulfate interfere in the calcium atomic absorption determination and 200 mg/liter of each caused a 35% and a 30% depression, respectively (Ref.: Parker, C. R., "Water Analysis by Atomic Absorption Spectroscopy", Varian Techtron, Palo Alto, California, 78 P., 1972) Both effects are masked by the addition of lanthanum chloride. Concentrations of more than 1,000 mg/liter magnesium and 500 mg/liter each of sodium and potassium cause a 5-10% enhancement due to suppression of the calcium ionization (Ref.: Brown, E., et al., Methods for Collection and Analysis of Water samples for Dissolved Minerals and Gases in Techniques of Water Resources Investigations of the U. S. Geological Survey, Chapter 1, U. S. Geological Survey, Washington, D. C., 1970, 160 P.). The magnesium determination is interfered with by more than 400 mg/liter each of sodium, potassium, and calcium, while phosphates and sulfates also cause interferences. Silicates and carbonates at 200 mg/liter each caused a 42% and 17% depression, respectively (Parker, 1972). Phosphate, for example, will bind the magnesium and prevent the magnesium atomic absorption when the light passes through the flame. Interferences in the calcium and magnesium determination are reduced by addition of an inorganic "releasing" agent, lanthanum, in a concentration of 10,000 mg/liter in the sample. This prevents the binding of calcium and magnesium by anions. The interferences are also reduced by using a nitrous oxide-acetylene flame instead of the lower temperature air-acetylene flame.

J. Hardness:

Method: The hardness reflects the total concentration of polyvalent metals, mainly calcium and magnesium; but will include iron, zinc, and copper when present in significant quantities. The most accurate method is by summation of the individual polyvalent metals, as measured by atomic absorption spectroscopy, which are then expressed as $CaCO_3$ equivalents.

A more rapid method is based upon ethylenediamine tetraacetic acid (EDTA) titration, using Eriochrome Balck T as indicator (Ref.: Standard Methods, pp. 178-184). This method measures the calcium and magnesium ions.

Interference:

Calculation from determination of individual cations: This method is subject to the usual interferences encountered in the atomic absorption determinations of the individual cations.

EDTA titration: Some metal ions interfere with this procedure by causing fading or indistinct endpoints. This interference is reduced by the addition of certain inhibitors to the water sample before titration with EDTA. The maximum concentrations of interfering substances which may be present in the original sample and still permit titration with EDTA are shown in Table 122(1), p. 180 of Standard Methods. Suspended or colloidal organic matter in the sample may also interfere with the endpoint but may be overcome by evaporating the aliquot to dryness on a steam bath followed by heating in a muffle furnace at 550°C until the organic matter is completely oxidized.

K. Heavy Metals:

Sample pretreatment for Metal Analysis:

In the EPA Manual, Methods for Chemical Analysis of Water and Wastes, 1974, metal content of a sample is divided into four categories as follows:

- Dissolved metals: those constituents (metals) which will pass through a 0.45 μ membrane filter.
- Suspended metals: those constituents (metals) which are retained by a 0.45 μ membrane filter.
- Total metals: the concentration of metals determined on an unfiltered sample following vigorous digestion or the sum of the concentrations of metals in both the dissolved and suspended fractions.
- Extractable metals: the concentration of metals in an unfiltered sample following treatment with hot dilute mineral acid.

Procedures are given for sample treatment prior to the determination of metals in the four categories described. This information is found in pp. 81-83 of the above reference. A decision should be made, prior to sampling and analysis, relative to what type of metal content information is required; i.e., dissolved, suspended, total or extractable, and the sample should be handled accordingly.

Method: Most heavy metals are determined by atomic absorption spectroscopy methods (EPA Manual, pp. 78-157). Standard additions are used for leachates of high strength to determine the magnitude of the interference. Standard additions should be used for the elements lead, copper, nickel, and chromium, but may be omitted for zinc and cadmium. For total metal analysis, the sample should be collected in a polyethylene bottle and acidified to pH 2 with 1:1 redistilled nitric acid. When the dissolved metals, those filterable through a 0.45 μ filter, are determined, the suspended metals should be determined concurrently. Mercury is determined by atomic absorption with the cold vapor technique (EPA, pp. 118-122).

Colorimetric methods are recommended for arsenic and selenium. Arsenic is determined with silver diethyldithiocarbamate (EPA pp. 9-10) and selenium is determined by the diaminobenzidine method (Standard Methods, pp. 296-298).

Interference: Several interferences are reported for the atomic absorption determination and can be classified as chemical, non-atomic, ionization and spectral (EPA, p. 84). Chemical interference occurs when heavy metals are not available in the atomic form because of molecular combination with anions present in the solution. This interference can sometimes be overcome by addition of a releasing agent such as lanthanum. The presence of a high concentration of dissolved solids may cause a non-atomic interference such as by light scattering. This effect is partially corrected by measuring absorbance of a nearby non-absorbing wavelength. Ionization interferences occur when the atom ionizes in the flame, after which it is not available for atomic absorption. This interference is reduced by addition of an easily ionized element such as cesium to suppress the ionization of the analyte atom. Spectral interference can occur when the absorbing wavelength of another element falls within the width of the absorption line of the analyte atom. Such interference is partially reduced by narrowing the slit width of the instrument.

The analysis of mercury by atomic absorption with the cold vapor technique depends on the reduction of the sample with $SnSO_4$ or $SnCl_2$, which may not be complete when other oxidants in high concentrations are present. Interference from certain volatile organic materials which will absorb at the wavelength of mercury (253.7nm) is also possible. A preliminary run without reagents should determine if this type of interference is present.

The determination of arsenic and selenium by atomic absorption spectroscopy using the gaseous hydride method may not be satisfactory, since reduction to the trivalent form with $SnCl_2$ may not be complete. The conversion to gaseous arsine after addition of zinc metal may also not be complete.

In the colorimetric determination of arsenic with silver diethyldithiocarbamate, chromium, cobalt, copper, mercury, molybdenum, nickel, platinum, and silver may interfere in the generation of arsine. High sulfur content of a sample may exceed removal capacity of the lead acetate scrubber. Samples should be spiked with a known amount of arsenic to establish adequate recovery.

For the colorimetric determination of selenium with diaminobenzidine, no inorganic compounds give a positive interference. Negative interference results from compounds that lower the concentration of diaminobenzidine by oxidizing this reagent. Iodide and, to a lesser extent, bromide, cause low results.

4. Biological Parameters:

A. Biochemical Oxygen Demand (B.O.D.)

Method: The biochemical oxygen demand determination is a bioassay-type procedure and measures the oxygen demand exerted by microorganisms during uptake of degradable substrates and by chemical oxidation reactions. The sample is incubated under aerobic conditions in the dark at 20°C during a five-day period.

The B.O.D. determination is run according to Standard Methods, pp. 489-495, using dilution water which is seeded with settled domestic sewage. B.O.D. values obtained should be judged carefully and be determined parallel with comparable chemical tests, such as free volatile fatty acids, C.O.D. or T.O.C.

Interference: Relatively low results are obtained with the test when toxic compounds are present that inhibit the bacterial population or when a biomass is used that is not adapted to the specific substrate.

B. Coliform Bacteria (Total and Fecal)

Method: The most probable number (MPN) technique should be selected for leachate monitoring purposes, as opposed to the membrane filter (MF) technique, since it is able to detect bacteria at lower concentrations and is less subject to suspended solids interference. Inactivation studies, however, in which a certain amount of bacteria is added to a sample to study its subsequent reduction with time, should be conducted if the MF technique is used. Presumptive and confirmed tests are run for total coliforms and the completed coliform test is run in those instances where leachate causes pollution of drinking water supplies. (Ref.: Standard Methods, pp. 664-668)

The fecal coliform MPN procedure is used as a confirmatory test procedure in conjunction with prior enrichment in a presumptive test medium for optimum recovery of fecal coliforms. (Ref.: Standard Methods, pp. 669-672)

Interference: The results of the examination with the multiple tube technique are expressed in terms of a statistical Most Probable Number (MPN) and the estimate generally tends to be greater than the actual numbers. An observed limitation of the membrane filter technique is its reduced ability to detect bacteria in turbid samples or in the presence of high concentrations of noncoliform bacteria.

5. Miscellaneous Determinations:

Some of the miscellaneous leachate parameters which have been given attention in various studies are: Methylene blue active substances, cyanide, fluoride, sulfide, silica, hexane solubles, ether solubles, color, visual appearance, and odor.

7.5.1 Additional Valuable Information

Additional valuable information on specific analytical methods is available in "Procedures for the Analysis of Landfill Leachate", in Proceedings of an International Seminar, Environmental Conservation Directorate, Ottawa, Ontario, Report EPS-4EC-75-2, October 1975.

7.6 FIELD TESTING VERSUS TESTING IN THE LABORATORY

The majority of tests performed on leachate samples are run in the analytical laboratory on samples which have been preserved by refrigeration or chemical means. A limited number of tests, however, can be performed at the sampling site on a freshly drawn sample. There are a number of advantages in field testing in which sample degradation is practically eliminated, along with the need for sample preservation, transportation, and handling. An added advantage is the ability to re-sample and re-analyze immediately, on site, if it is suspected that a particular sample is not representative or valid. There are also disadvantages encountered in field testing and these usually relate to the reliability of the particular method and equipment used for the test.

Some tests can be run in the field with the same methods and equipment which would be used in the laboratory and yield the same reliability. Among such tests are those involving the measurements of pH, oxidation reduction potential, specific conductance, turbidity, dissolved oxygen, and specific ions by means of specific ion electrodes. The equipment used in these tests is available in portable models which are of equal applicability in the field and laboratory.

Other tests are sometimes performed exclusively in the field using methods and equipment specifically designed for field use. A number of commercial kits are available for such purposes.

While offering distinct advantages, there are also disadvantages inherent in the use of field kits. The following evaluation of field kit useage is given in Handbook for Monitoring Industrial Wastewater, U. S. EPA, Technology Transfer, August 1973, p. 5-14:

> Estimating the Amount of Pollutants Present by Use of "Kits"
>
> Companies, such as the Hach Chemical Company, Delta Scientific, Inc., and Koslow Scientific Company have

> manufactured "Kits" for the analysis of various constituents of wastewater. The kits consist of a small portable container in which all the necessary equipment and instructions are conveniently packaged and arranged to perform a variety of tests. No previous laboratory training is required and, within minutes, an indication of the chemical constituents in wastewater can be determined.
>
> Koslow Scientific and the Hach Company provide kits for determining the presence of heavy metals, such as Cd, Hg, and Pb, and includes reagents for masking interferences.
>
> The major disadvantage in using kits is the inability of the pre-packaged devices and reagents to effectively cope with interferences. Reference 2 (Standard Methods for the Examination of Water and Wastewater, 13th Edition, American Public Health Association, 1971) outlines procedures for the removal of interferences by pretreatment techniques and the reagents necessary for masking these interferences that are usually not available in the kits. The accuracy of the tests performed with kits is usually less than that obtainable with precise laboratory techniques. Kits give good results in relatively clean water but pose problems when used to analyze wastewaters. They are nevertheless useful in preliminary surveys performed to determine overall characteristics of a wastewater.

This evaluation of the use of field kits for the analysis of industrial wastewater is equally applicable in the case of leachate analysis. Due to their limited accuracy, the use of kits may not be acceptable for most enforcement cases.

7.7 MOBILE LABORATORIES

Although not in widespread usage, mobile analytical laboratories have the potential of providing a combination of laboratory capability and field-testing convenience. The instrumentation and general capability of a mobile laboratory can vary over a wide range, depending upon its application, manpower, and the capital investment involved. By using normal laboratory equipment and methods, the mobile laboratory can obtain results equivalent to those of a conventional analytical laboratory, while incorporating all of the advantages of field kits. Limitations imposed by sample degradability and work load will be encountered by the mobile laboratory in much the same way as experienced by the conventional laboratory under certain conditions. If a sample or samples presented to a mobile laboratory must be analyzed for a large number of parameters (i.e., 20 or 30), then sample degradation versus work load will have to be addressed. The sample will have to be preserved and the analyses prioritized relative to order of degradability. In this respect, the mobile laboratory shares the disadvantages of the conventional laboratory, along with its advantages.

7.8 AUTOMATED METHODS

Automated wet chemistry methods offer many advantages, among which are economy, increased precision, and accuracy when applied to repetitive analytical work loads of significant volumes. Federal, state and local regulatory agencies, industry, educational institutions, and independent testing laboratories (among others) use automated methods to handle large, demanding repetitive analytical work loads.

Automated wet chemistry is addressed in the Handbook for Monitoring Industrial Wastewater, U. S. EPA, August 1973, p. 5-14 as follows:

> Automated wet chemistry is frequently used in analysis of wastewaters and for automated monitoring of waste effluents. When used, the system consists of a sampler to select air, reagents, diluents, and filtered samples. From the sampler, the fluids pass through a proportioning pump and manifold where the fluids are aspirated, proportioned, and mixed. The samples are then ready for separation by passing through any one of the following units: a dialyzer (continuously separates interfering materials in the reaction mixture); a digester (used for digestion, distillation, or solvent evaporation); a continuous filter (for on-stream separation of particulate matter by a moving belt of filter paper), or a distillation head (separates high vapor pressure components).
>
> After separation, the samples can be conditioned in a constant temperature heating bath. After conditioning, the samples pass through a detection system which may be a colorimeter, a flame photometer, a fluorometer, a UV spectrophotometer, an IR spectrophotometer, an atomic absorption spectrophotometer, or a dual differential colorimeter. The signals from the detection system are sent to a recorder or a computer system.

In the May 1975 issue of Environmental Newsletter, a publication of Technicon Industrial Systems, a list of water quality major automated methods is presented (Table 15). It should be noted that ten of the methods are Federal Register approved and eleven of the methods are presented in the U. S. EPA Manual, Methods for Chemical Analysis of Water and Wastes, 1974.

Automated methods are discussed in Standard Methods for the Examination of Water and Wastewater, 13th Edition, 1971, pp. 14-15, as follows:

Parameter	Technicon or Other Method	Federal Register Approved	Individual Variance Approvals	1974 EPA Methods Book Reference (1)	Practical Method (2)	Usual Method Range (3)
Acidity (Thymol Blue)	164-71W				X	0-500mg/l
Alkalinity (Methyl Orange)	111-71W	X		P. 5		0-500mg/l
Aluminum	(4)				X	
Ammonia (Dialysis)	270-73W				X	0-1mg/l
Boron	202-72W				X	0-1mg/l
Chloride	**99-70W**	X		**P. 31**		**0-10mg/l**
Chromium (Hexavalent)	**162-71W**				X	**0-0.2mg/l**
COD	**137-71W**				X	**0-100mg/l**
Color	**181-72W**				X	**0-250 Units**
Copper	**211-72W**				X	**0-2mg/l**
Cyanide	315-74W				X	0-500 μg/l
Fluoride	129-71W			P. 61	X	0-2mg/l
Hardness (Total)	165-71W	X		P. 70		0-250mg/l
Iron	109-71W		X		X	0-1mg/l (ppm)
Mercury	– – –			P. 127		0-20 μg/l
Nitrate & Nitrite (Dialysis)	**271-73W**				X	**0-1mg/l**
Nitrogen (Ammonia)	**98-70W**	X		**P. 168**		**0-10mg/l**
Nitrogen (Kjeldahl, Total)	**146-70W**	X		**P. 182 (HgO)** **P. 190 (SeO_2)**		**0-10mg/l**
Nitrogen (Nitrite)	**102-70W**	X				**0-1mg/l (N)**
Nitrogen (Nitrate & Nitrite)	**100-70W**	X				**0-2mg/l (N)**
Nitrogen (Organic plus Ammonia)	325-74W(5)		X			0-0.40mg/l or 0-10mg/l
NTA	– – –			P. 220		0-10mg/l
Phenol	127-71W		X	P. 243		0-500 μg/l
Phosphorus (Total)	94-70W(6)	X				0-10mg/l
Phosphorus (Total)	297-73W 188-72W			P. 256	X	0-10mg/l 0-1mg/l
Phosphorus (O-phosphate)	**[illegible]-70W**	X				**0-10mg/l**
[illegible]	**[illegible]-71W**				X	**0-10mg/l**
[illegible]	**[illegible]-73W**				X	**0-1000mg/l**
	[illegible]-73W				X	**0-100mg/l**
[illegible]	**118-71W**	X		**P. 279**		**0-300mg/l**
[illegible]	**[illegible]-72W**				X	**0-3mg/l**

1) Some EPA Methods may differ in detail from the Technicon Listed Method. In such cases the method to be selected for use is determined by review of sample matrices and ranges required.

2) Method in practical use but general regulatory approval not yet obtained.

3) Method ranges can be adjusted to suit particular needs. Method resolution is typically 1% of full scale.

4) Norwegian Institute for Water Research, Oslo, Norway, A. Henderson

5) Manual digestion followed by autoanalysis for ammoniacal nitrogen.

6) Manual digestion followed by autoanalysis for phosphorus.

TABLE 15. CLASSIFICATION OF MAJOR METHODS FOR AUTOANALYZER I AND II SYSTEMS

(Technicon Industrial Systems, Environmental News Letter, May, 1975)

Automated analytical instrumentation: Automated analytical instruments are now available and in use to run individual samples at rates of 10 to 60 samples per hour. The same instruments can be modified to perform analyses for two to twelve constituents simultaneously from one sample. The instruments are composed of a group of interchangeable modules joined together in series by a tubing system. Each module performs the individual operations of filtering, heating, digesting, time delay, color sensing, etc. that the procedure requires.

The read-out system employs sensing elements with indicators, alarms, and/or recorders. For monitoring applications, automatic standardization-compensation, electrical and chemical, is done by a self-adjusting recorder when known chemical standards are sent periodically through the same analysis train. Such instrument systems are presently available.

Appropriate methodology is supplied by the manufacturer for many of the common constituents of water and wastewater. Some methods are based on procedures described in this manual while others originate from the manufacturer's adaptation of published research. Since a number of methods of varying reliability may be available for a single constituent of water and wastewater, a critical appraisal of the method adopted is obviously mandatory.

Automated methodology is susceptible to the same interferences as the original method from which it derives. For this reason, new methods developed for automated analysis must be subjected to the exacting tests for accuracy and freedom from adverse response already met by the accepted standard methods.

Off color and turbidity produced during the course of an analysis will be visible to an analyst manually performing a given determination and the result will be properly discarded. Such abnormal effects caused by unsuspected interferences might escape notice in an automated analysis. Calibration of the instrument system at least once each day with standards containing interferences of known concentration could help to expose such difficulties. Routine practice is to check instrument action and guard against questionable results by the insertion of standards and blanks at regular intervals--perhaps after every 10 samples in the train. Another important precaution is proper sample identification by arrangement into convenient groups.

> In brief, a fair degree of operator skill and knowledge, together with adequately detailed instructions, is required for successful automated analysis.

In the report "Compilation of Methodology for Measuring Pollution Parameters of Landfill Leachate" by E. S. K. Chian and F. B. DeWalle, the following comments are made concerning automated methods:

> All automated methods as recommended by EPA (1974) for water and wastewater and Technicon (1973) for industrial waste should be evaluated for possible interferences since most tests are based on colorimetric analyses which are generally subject to strong interferences by the color and suspended solids present in leachate. Such evaluation is necessary since increasing amounts of leachate samples will be analyzed by automated methods at a future date.

7.9 LABORATORY QUALITY CONTROL

The subject of laboratory quality control is treated in detail in Handbook for Analytical Quality Control in Water and Wastewater Laboratories, U. S. EPA Technology Transfer, June, 1972. The various topics covered include: Importance of Quality Control, Laboratory Services, Instrumental Quality Control, Glassware, Reagents, Solvents and Gases, Control of Analytical Performance, Data Handling and Reporting, Special Requirements for Trace Organic Analysis, and Skills and Training. A number of valuable references are provided in each section.

The technical and legal aspects of an adequate quality control program are of prime importance in the analysis of sanitary landfill leachate samples. The investment of time and effort needed for a quality control program are well compensated in the resultant reliability of and confidence in the data obtained.

The economics of quality control is greatly favored in the use of automated analysis systems as compared to manual systems. In a recent issue of U. S. EPA's Analytical Quality Control Newsletter (October, 1975, p. 5), it is stated that for a particular automated system, the additional personnel work load required to provide an analytical quality assurance overhead of 40% is estimated to be about 1%. The 40 to 1 advantage is most impressive.

7.10 MANPOWER AND SKILL REQUIREMENTS

Manpower and skill requirements for analytical work are dependent upon a number of factors, including nature of the sample, work load, analytical parameter to be tested, method used, sensitivity, precision and accuracy desired, and equipment and facilities available. A consideration of major importance, of course, is whether the analyses will be performed by manual or automated methods.

In *Handbook For Analytical Quality Control In Water and Wastewater Laboratories*, U. S. EPA. June 1972, pp. 9-2, 9-3, 9-4, skills and skill-time ratings for standard manual analytical operations are discussed in detail.

Manpower and skill requirements are reduced dramatically when automated methods are used. The usual skill requirements are those of a technician for preparation of samples, solutions, calibration, and glassware handling. Automated data processing affords additional manpower savings.

A comparison of manual versus automated analytical methods for throughput, space, and personnel requirements is given in "Automated Methods For Assessing Water Quality Come of Age", by M. J. F. Du Cros and J. Salpeter, *In* Environmental Science and Technology, Vol. 9, Number 10, October 1975, p. 932.

Data are presented in tabular form for the analysis of 12 water quality parameters. For the manual methods, 12 personnel and 114-feet of bench space are required to perform 100 to 126 determinations per hour. For the automated methods, 3 personnel and 45-feet of bench space are required to perform 370 determinations per hour. In addition to an appreciable savings of space, the automated throughput is approximately 12 times that achieved with manual methods.

7.11 RECORDS, DATA HANDLING, AND REPORTING

A significant amount of analytical data are generated in a leachate testing program. The data must be handled, interpreted, checked for validity, recorded, and reported. This is an important aspect of the testing program and should be given appropriate attention. If the data are not properly handled, the considerable effort and expense involved in sampling and analysis can be lost or erroneously applied. It should be noted that legal as well as technical considerations can be associated with records, data handling, and reporting.

A detailed discussion on data handling can be found in "Data Handling and Reporting", in Handbook for Analytical Quality Control in Water and Wastewater Laboratories, U. S. EPA, June 1972, pp. 7-1 to 7-11. Amoung the topics treated in this chapter are: Significant Figures, Accuracy Data, Precision Data, Report Forms, Digital Read-Out, Key Punch Cards and Paper Tape, STORET- Computerized Storage and Retrieval of Water Quality Data, and SHAVES- a Consolidated Data Reporting and Evaluation System.

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